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CORAL REEFS OF MISKITUS CAYS, NICARAGUA

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ABSTRACT: The Miskitus Cays, on the Caribbean coast of Nicaragua, consist of eighty mangrove and two sand and gravel cays, surrounded by seagrass beds, octocoral gardens, patch reefs, reef crests, extended algae platforms, short reef walls, and two marginal reefs around the sand cays. Seventy sites were inspected and eighteen sites were selected for rapid assessments in order to determine the status of the coral reefs. Linear transects and the intercept point methods were used to determine the relative benthic cover, and the density, size and health of coral colonies was estimated following the AGRRA protocol. Water was highly turbid due to the shallowness of the reefs and high wave energy. Northwest reefs, closer to the Coco river mouth, were affected by terrestrial sediments and were overgrown by algae whereas storm damage was evident in the eastern reef crest fronts. In total, 39 stony coral species were found and 12 new species were reported for Cayos Miskitus. Mean live coral cover was high (43.4%), but it was still lower than the algae cover (54.2%). Mean coral diameter (59.7 cm) and height (4.2 cm) were high but total mortality (27.9%), bleaching (4%) and diseases (3%) were low. Reefs of Nicaragua are in the best condition of the Caribbean region of Central America but good management of the fisheries, the marine reserve, and the Coco river basin are urgent to maintain reef quality.

INTRODUCTION

The Caribbean shoreline of Nicaragua is about 463 km long with a broad continental shelf where coral reefs grow, especially in the north section (Murray et al. 1982, Hallock et al. 1988). The coast is divided into the Autonomous North Atlantic Region (RAAN) and the Autonomous South Atlantic Region (RAAS); the RAAN is mainly inhabited by the Miskitus indigenous people (USAID 1996).

Coral reefs of the Caribbean coast of Latin America are described in Cortés (2003). The Miskitus Cays Biological Reserve, created in 1991, is located 50 km from the coast, northeast from Puerto Cabezas inside the RAAN. It has a radius of 40 km around Major Miskitu Cay (14°23'N-82°46'W) totaling 50,000 ha, and a coastal and marine belt 20 km wide between Wounta and Gracias a Dios Cape. The total area of the reserve is 765,867 ha (Jameson 1996). The marine ecosystem consists of eighty mangrove and two sand and gravel cays, surrounded by a mosaic of shallow interconnected marine habitats (Alevizon 1993, Jameson 1996). The communities in the north littoral of Nicaragua, mainly Sandy Bay and Puerto Cabezas, use the area of this marine reserve for artisan fishing. Fishing for shrimp and finfish is done close to the coast, whereas lobster, shark and turtle fishing is concentrated around the cays (Harrington and Gallucci 1996, USAID 1996, Maradiaga 1998).

The first scientific study in Miskitus Cays was on algae and seagrass (Phillips et al. 1982), followed by Ogden and Gladfelter (1983) and Marshall (1984). Between 1993 and 1995 the "Ministerio del Ambiente y los Recursos Naturales de Nicaragua (MARENA)", the Caribbean Conservation Corporation (CCC) and the United States Agency for International Development (USAID) developed a preliminary management plan for the reserve where a general description of coral reefs, seagrasses and lagoons was provided. Alevizon (1993) provided qualitative descriptions of these coral reefs and their fishing resources and Jameson (1996) reported 27 coral species from this system. There are no quantitative studies in this section of the Caribbean coast of Nicaragua, although the coral reefs of Corn Island in the RAAS have been widely studied (reviewed in Ryan and Zapata 2003).

The main objective of this study was to quantitatively assess the coral reef status in Miskitus Cays in comparison with other reefs in the Caribbean coast of Central America, providing baseline data that can be used for future research and monitoring.

MATERIALS AND METHODS

A rapid reef assessment was conducted in August 2001 inside the 50,000 ha around the Major Miskitu Cay (14°23'N-82°46'W; Figure 1) in order to diagnose the status of its coral reefs in terms of reef substrate cover, coral colony density, and health. In total, seventy sites recorded with GPS were inspected by snorkeling for 15 min to determine habitat type and coral species richness. These sites were assessed and distributed as follows: four to the north, seventeen to the northeast, two to the east, six to the southeast, three to the south, three to the southeast, ten to the west, and twenty five to the northwest. Of these, eighteen sites were selected for SCUBA diving and evaluation with linear transects. These were distributed as follows: one to the north, five to the northeast, one to the east, one to the southeast, three to the south, two to the southwest, two to the west, and three to the northwest. We report the site names used by the local fishermen.

Coral reefs are very shallow in this area, so three 10 m long

linear transects parallel to the depth contour and separated by 5 m each were evaluated in each site at a depth between 1 and 5 m. In sites with distinguishable reef crests (crashing waves), transects were done at the base of the crest. Generally, there were no reef platforms associated with these reef crests. Reef crest complexity was estimated in Creole Bar, Toro Cay and Witties by following the contour of the bottom with a chain (Rogers et al. 1994). In sites consisting of reef patches or reef spurs, transects were done over these formations.

Along the 10 m transects, the point intercept method was used to record the relative substrate cover every 25 cm (Nadon and Stirling 2006). Corals were identified after Humann (1993), and the AGRRA protocol (Lang 2003) was used to describe adult coral colonies longer than 25 cm, including their diameter (cm), height (cm), old and recent mortality, and presence of diseases. Analysis of Variance (ANOVA) followed by Bonferroni pairwise comparisons were used to compare the relative substrate cover (%) and coral colonies density (# colonies/10 m) among sites after log_{10} transformation of the data to aid in meeting the normality and homogeneity of variance assumptions.

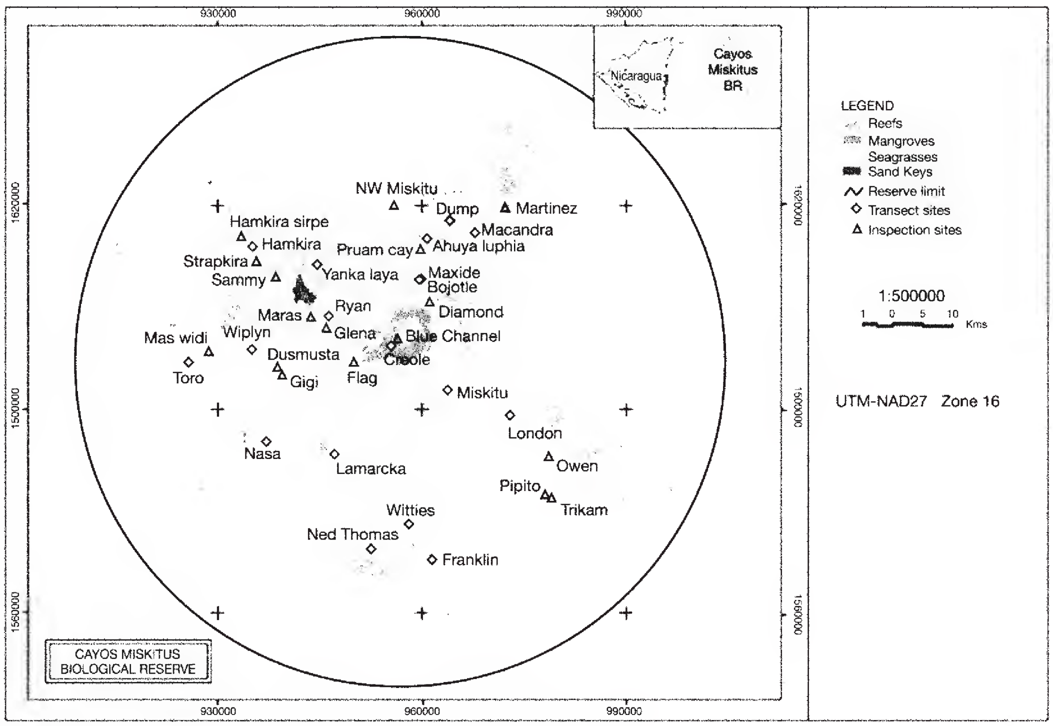

Figure 1.
Map of Miskitus Cays, Nicaragua. Transect sites have quantitative information whereas inspection sites have qualitative information (Modified from Valerio 2001). + = latitude and longitude grid points.

Results

Description of the reef habitats

The Miskitus Cays reef system is shallow (< 5 m) and consists of a series of mangrove cays (10 %), two sand and coralline gravel cays surrounded by a mosaic of extensive seagrass beds (73%) and algae, several octocoral gardens, patch reefs, and reef crests (17%) oriented from north to south. The reef system extends up to 20 km from the cays, and water visibility was low (5 to 10 m).

Seagrasses were dominated by *Thalassia testudinum* and/or *Syringodium filiforme* in sandy substrate, and they were found mainly around the mangrove cays and to both sides of the reef crests. Octocoral gardens, in sand patches or rocky bottoms, were dominated by *Pseudopterogorgia americana, Plexaura flexuosa, Gorgonia ventalina*, and some disperse colonies of live coral like *Diploria strigosa, Siderastrea siderea*, and *Porites astreoides*.

Reef crests had a mean complexity index of 1.84 at a depth between 1 to 5 m. Reef crests were formed mainly by *Acropora palmata*, most of which were dead, and *Millepora complanata*. Other dominant species were *Montastraea faveolata, S. siderea, D. strigosa, P. asteroides, Agaricia agaricites*, and *Agaracia tenuifolia*.

The windward base of the reef was up to 5 m deep and several reef bases were dominated by gardens of *Acropora cervicornis, Acropora prolifera* and big colonies (up to 2 m wide and high) of *M. faveolata*. In some sectors, the reef crests were adjacent to wide shallow platforms of flat carbonate rock covered by fleshy non-coralline algae, especially *Dictyota* spp., *Padina* spp., *Galaxaura comans, Sargassum natans, Asparagopsis taxiformis, Turbinaria turbinata* and *Stypopodium zonale*, with a few disperse coral colonies of *P. astreoides, D. strigosa*, and *S. siderea*. In other sectors, the reef fronts consisted of sand with seagrass beds or octocoral gardens also with low coral cover. In some reefs there were also deep walls (50 m) covered by a great diversity of reef organisms, small coral colonies and big octocorals and antipatharians, and in a few cases spurs and grooves were found at the base of these walls. Some of these crests presented an edge relatively close to the crest, with a small wall down to 16 m maximum, with complex caverns and great diversity, adjacent to sand and in some cases the seagrass *Halophila baillonis*. In the leeward side of the reef crests the substrate was usually covered by dead coral gravel, algae, octocorals and disperse colonies of

TABLE 1. *Damage by site and coral species (see Figure 1). Sites not listed had no damage. Codes: Black Spot Disease (BSD), White Band Disease (WBD), White Plague (WP), Black Band Disease (BBD) and White Spots (WS). Tumors = neoplasm; Damselfish chimneys = produced by damselfish bites.*

Sampling site	Coral species (Damage)
Bojotle Kira	*Siderastrea siderea* (BSD) *Montastraea faveolata* (BSD) *Acropora cervicornis* (WBD)
Ahuya Luphia	*Siderastrea siderea* (WP, BSD) *Acropora cervicornis* (WBD)
Martínez reef	*Siderastrea siderea* (BSD)
Dump	*Siderastrea siderea* (BSD)
Macandra	*Diploria strigosa* (Tumours) *Acropora palmata* (Damselfish chimneys)
Hamkira	*Acropora palmata* (Damselfish chimneys) *Acropora cervicornis* (WBD) *Montastraea faveolata* (BBD)
Wiplyn	*Siderastrea siderea* (BBD) *Acropora prolifera* (WBD) *Diploria strigosa* (WP)
Gigi	*Montastraea franksi* (BSD) *Montastraea faveolata* (BSD) *Siderastrea siderea* (WP, WS) *Colpophyllia. natans* (BBD)
Dusmusta	*Diploria strigosa* (BBD, Tumours, WP) *Diploria clivosa* (BBD) *Acropora palmata* (WBD) *Montastraea faveolata* (BBD)
Lamarcka reef	*Acropora palmata* (WBD)
Nasa reef	*Acropora cervicornis* (WBD) *Siderastrea siderea* (BSD) *Diploria strigosa* (BBD)
Witties	*Acropora cervicornis* (WBD) *Siderastrea siderea* (BSD)
Creole Bar	*Montastraea faveolata* (WP, BBD) *Siderastrea siderea* (BBD) *Colpophyllia natans* (BSD) *Montastraea annularis* (BSD)
Toro Cay	*Acropora cervicornis* (WBD) *Acropora palmata* (Damselfish chimneys)
Franklin reef	*Diploria strigosa* (BBD)
Ned Thomas	*Acropora prolifera* (WBD) *Siderastrea siderea* (BSD) *Acropora palmata* (Damselfish chimneys)
Yanka Laya	*Acropora palmata* (Tumours)

the corals *Siderastrea* spp., *Diploria* spp., and *Porites* spp.

Circular patch reefs, ranging from 20 to 100 m in diameter, were dispersed around the islands and were < 10 m deep. These reefs were also surrounded by sand patches and seagrass beds. There are a great number of patch reefs in the area around the cays, especially to the north near Ahuya Luphia and Bojotle Kira, to the northeast near Morrison Dennis cays (MARAS), and near Sammy, Strap Kira and Hamkira cays. Finally, there were two marginal reefs around the two sand cays in the northeast Hamkira and Hamkira sirpe. In these reefs, it was also possible to find healthy gardens of *A. palmata*, *A. prolifera*, and *A. cervicornis*.

In the west (Toro Cay, Glena Bar) and northwest (Sammy, reef patches around Hamkira) areas, significant suspended sediment was present which reduced water visibility. In this sector and some parts of Nasa and Gigi in the southwest, several reef patches were covered by a thick mat of the green algae, *Chaetomorpha gracilis*, and most coral colonies were dead. In Toro Cay and Glena Bar to the west, there was also a high cover of the cyanobacteria, *Schizotrix* spp.

Based on qualitative observations, diseased coral colonies were found in 24% of the inspected sites (Table 1). This is especially true at Wiplyn, Dusmusta and Gigi (the west), at Nasa reef (southwest), at Ned Thomas (south), at Creole Bar (north), and at Ahuya and Bojotle Kira (northeast). The most frequent disease in Miskitus cays was Black Band Disease (BBD) in colonies of *M. faveolata*, *S. siderea*, *Colpophyllia natans*, and *Diploria* spp. Furthermore, there was a great incidence of Black Spot Disease (BSD) in colonies of *S. siderea*, White Band Disease (WBD) in colonies of *Acropora* spp. and White Plague (WP) in colonies of *S. siderea*, *D. strigosa*, and *M. faveolata*.

Composition, richness, density and status of coral colonies

Thirty nine scleractinian coral species were found at Miskitus Cay (Table 2), similar to what has been found in other reefs from the southern Caribbean coast of Central America (Table 3). The richest sites were those in which the reef crest was associated with a vertical complex and/or deep wall (Table 4). These deep wall systems were Creole Bar, North West Miskitu reef, and Macandra and Witties in the south. Nassa reef in the southeast also exhibited high coral species richness but had a short wall down to 10 m deep. Finally, the patch reefs of Ahuya Luphia were also rich due to their 3 to 8 m high depth. The poorest sites in coral species richness were the patch reefs located in carbonate rock platforms covered with fleshy macro-algae such as Martínez reef, Owen shoal and Flag Reef. The poorest reef crest sites were Yanka laya, Ryan and Sammy in the northwest, Mas Widi in the west, Trikam rock in the southeast and Maxide and Dump in the northeast.

Coral colony density was significantly different among sites ($F_{17,36} = 2.17$, $p = 0.025$). Sites with densities between 1 and 10 colonies per 10 m were patch reefs such as, Bojotle Kira and Ahuya Luphia in the north, and Ryan, one of the most deteriorated reef crests in the northwest and Maxide to the northeast. Density was > 16 colonies in Macandra and Creole bar to the north, Franklin reef and Ned Thomas in the south, Nasa reef and Toro cay in the southwest, and Hamkira and Yanka Laya in the northwest (Table 5).

The dominant (> 35%) coral species in most of the reef crests were *A. palmata* and *M. faveolata* with exception of the

TABLE 2. *List of marine invertebrates observed in Miskitus Cays, Nicaragua coral reefs in August 2001.*

SPONGES (Phylum Porifera, Class Demospongiae)
Callyspongia plicifera
Ircinia strobilina
Diplastrella megastellata
Agelas confiera
Cliona sp.

MILLEPORINES (Phylum Cnidaria, Class Hydrozoa, Order Milleporina)
Millepora alcicornis
Millepora complanata

ANEMONES (Phylum Cnidaria, Class Anthozoa, Order Actinaria)
Stichodactyla helianthus
Condylactis gigantea
Bartholomea annulata

ZOANTHIDS (Phylum Cnidaria, Class Anthozoa, Order Zoanthidea)
Palythoa caribaeorum
Zoanthus pulchelus

SCLERACTINIANS (Phylum Cnidaria, Class Anthozoa, Order Scleractinia)
Acropora cervicornis
Acropora palmata
Acropora prolifera
Porites porites
Porites astreoides
Oculina diffusa
Madracis mirabilis
Madracis decactis
Stephanocoenia michelinii
Montastraea annularis
Montastraea faveolata
Montastraea franksi
Montastraea cavernosa
Dichocoenia stokesii
Favia fragum
Siderastrea siderea
Siderastrea radians
Solenastrea bournoni
Solenastrea hyades
Diploria strigosa
Diploria clivosa
Colpophyllia natans
Meandrina meandrites
Manicina areolata
Leptoseris cucullata
Agaricia grahamae
Agaricia agaricites
Agaricia tenuifolia
Mycetophyllia danaana
Mycetophyllia lamarckiana
Mycetophyllia aliciae
Mycetophyllia ferox
Isophyllastrea rigida
Scolymia cubensis
Scolymia lacera
Mussa angulosa
Eusmilia fastigiata

CTENOPHORES (Phylum Ctenophora, Class Tentaculata)
Leucothea multicornis
Ocyropsis crystallina

SEA CUCUMBERS (Phylum Echinodermata, Class Holothuroidea)
Isostichopus badionotus
Holothuria mexicana

SEA URCHIN (Phylum Echinodermata, Class Echinoidea)
Echinometra viridis
Echinometra lucunter
Diadema antillarum
Meoma ventricosa

SEA STARS (Phylum Echinodermata, Class Asteroidea)
Oreaster reticulatus
Linckia guildingii
Asterina folium

TUNICATES (Phylum Chordata, Class Ascidiacea)
Clavelina puertosecensis

sites where the majority of *A. palmata* colonies were dead such as Witties, where *P. astreoides* (45%) dominated. Wiplyn was dominated by *D. strigosa* (40%) and *A. prolifera* (40%), Toro Cay was dominated by *A. tenuifolia* (48%), the fringing reef of Hamkira was dominated by *A. cervicornis* (42%), and Ryan crest and Bojotle Kira reef patches were dominated by *S. siderea* (30% and 50%, respectively). Only in the spurs of Creole Bar and the deep reef patches of Ahuya Luphia was *M. annularis* abundant (33% and 20%, respectively). *Montastraea faveolata* was especially abundant in Nasa reef (76%), *D. strigosa* in London reef (100%), and *A. palmata* in Yanka Laya (79%). The fire coral, *Millepora complanata*, was also common in these reefs and it contributed to the construction of the reef crests.

In general, the mean diameter and height of coral colonies was high in Yanka Laya, Toro Cay, Nasa reef and Lamarcka. The lowest mean diameter was found in London reef. The mean height was also higher in Maxide and Lamarcka (Table 6). The largest colonies of *A. palmata* (> 80 cm) were found in Miskitu reef, Maxide, Ned Thomas, Lamarcka, Hamkira and Yanka Laya, whereas relatively small colonies (diameter < 50 cm) were noted in Ryan. The colonies of *D. strigosa* were larger (> 50 cm) on average in Franklin reef, Macandra and Bolotle Kira, whereas those of *M. faveolata* > 1 m diameter were found in Macandra and Ryan. *Siderastea siderea* was larger (50 cm) in Wiplyn and Ryan (Table 7) than other areas. Overall, the mean diameter and height of coral colonies for all the reef system of Miskitus Cays was high (59.7 cm and 46.2 cm, respectively).

Recent mortality was 3.2%, old mortality was 25.7% and total mortality 28.9%. The ratio between live and dead coral was high (3.9, Table 6). Recent mortality of coral colonies was highest in Witties (24%), while old mortality was highest in Nasa reef (54%), followed by Ned Thomas (49%) and Bojotle Kira (49%) (Table 6). Total mortal-

TABLE 3. *Comparison of stony coral species richness in the Central American Caribbean coast.*

Country	Site	Reference	# Coral species
Panamá	Bocas del Toro	Guzmán 1998	32
Costa Rica	Cahuita	Cortés 1996-1997	41
	Manzanillo	Cortés 1996-1997	32
Nicaragua	Miskitu Cays	This study	39
	Corn Island	Ryan and Zapata 2003	25
Honduras	Cayos Cochinos	Guzmán 1998	56
	Roatán	Siirila 1992, Villeda et al. 1997	47
Guatemala	Punta Manabique	Fonseca 2003	29
Belize	Carrie Bow Cay	Cairns 1982	44

ity was highest in Nasa reef (54%) followed by Bojotle Kira (50%). The ratio between live and dead colonies was highest in Hamkira and lowest in Nasa reef. Bleaching was highest in Ahuya Luphia (20%) and Creole Bar (19%). Mean bleaching for Miskitus Cays was 4% (Table 6) and in only 6 of 18 sites (33%) was there evidence of bleaching, and it was always partial. I found disease in only 28% of the sites (5 of 18 sites) and in 26% of the coral species (10 of 39 species). The highest incidence of disease was found in Wiplyn (40%) and mean disease incidence overall was 3% (Table 6). There was not evident damage by anchors, but damage by storms was great, since colonies of *A. palmata* were fragmented and dead, especially in all the reef fronts and mainly in the ones to the east of Miskitus Cays.

Relative substrate cover analysis

There was a significant difference among sites and low variability within each site on coral live cover ($F_{17,36} = 2.399$, $p = 0.01$) and algae ($F_{17,36} = 2.238$, $p = 0.02$). The sites with higher live coral cover (> 50%) than the algae cover were the reef crests of Dump in the north, Franklin reef and Ned Thomas in the south, Lamarcka and Nasa reef in the southwest, and Hamkira and Yanka Laya in the northwest. The other sites had higher algae cover than live coral cover. The patch reefs of Bojotle Kira were the sites with less live coral cover and higher algae cover, followed by the reef patches of Ahuya Luphia, and the reef crest of Ryan. However, only the patches of Bojotle Kira had significantly lower coral cover than all other sites after Bonferroni pairwise comparisons ($p < 0.05$). Moreover, in Ahuya Luphia, the bottom consisted mainly of sandy material within the colonies, whereas in Ryan it consisted mainly of mud. Mean live coral cover for Miskitus Cays was high (43.4%), but lower than the algae cover (54.2%). In all sites, the percentage of non-coralline algae was higher than the percentage of coralline algae. In Miskitu reef (or Farrel reef) in the east, London reef in the southeast, Witties in the south and Wiplyn in the west a higher dead coral cover with algae than live coral cover was found (Table 8).

TABLE 4. *Number of coral species ranked by site from August 2001 collections.*

Site	***Physical Attribute***	***# coral species***
Creole Bar-Blue Channel	Reef Crest/Deep Wall	31
Macandra	Short Wall	29
Witties	Reef Crest/Short Wall	29
Ahuya Luphia	Patch Reef	28
NW reef	Deep Wall	27
Nassa	Reef Crest/Short Wall	25
Ned Thomas	Reef Crest	20
Franklin reef	Reef Crest/Deep Wall	17
Hamkira	Fringing Reef	17
Gigi	Reef Crest	16
Toro cay	Reef Crest/Short Wall	16
Hamkira sirpe	Fringing Reef	16
London reef	Reef Crest	15
Wiplyn	Reef Crest	15
Bojotle Kira	Patch Reef	14
Lamarcka	Reef Crest	14
Dusmusta	Reef Crest	14
Miskitu reef - Farrel reef	Reef Crest	13
Glena Bar	Reef Crest	13
Maxide	Reef Crest	11
Dump	Reef Crest	11
Sammy	Reef Crest	11
Ryan	Reef Crest	11
Yanka Laya	Reef Crest	8
Mass Widi	Reef Crest	7
Martínez reef	Patch reef	7
Flag reef	Patch reef	7
Owen shoal	Patch reef	5
Trikam rock	Reef crest	3

TABLE 5. *Density (mean ± standard deviation) of coral colonies (n=3) by site along transects from August 2001 collections.*

Site	**Density (Colonies/10 m transect)**
Maxide*	4.0 ± 1.0
Bojotle Kira*	4.7 ± 2.5
Ryan	10.0 ± 7.8
London reef	11.7 ± 6.4
Ahuya Luphia*	13.7 ± 5.5
Dump	13.7 ± 7.0
Miskitu reef	14.0 ± 5.6
Wiplyn	14.7 ± 9.0
Lamarcka	15.0 ± 1.7
Witties	15.7 ± 1.5
Toro cay	16.7 ± 7.1
Creole Bar	17.7 ± 1.2
Ned Thomas	18.3 ± 5.0
Macandra	18.3 ± 5.7
Nasa reef	18.7 ± 2.3
Hamkira	20.0 ± 11.4
Franklin reef	20.7 ± 6.0
Yanka Laya	21.3 ± 4.0

* Counted only colonies with a diameter > 25 cm.

Discussion

Miskitus Cays show a great diversity of interconnected marine habitats and resources which give shelter to reef species of great commercial value like turtles, dolphins, sharks, reef fishes, lobsters and queen conchs which are the base of the economy of the local fishing communities (Ryan et al. 1998). Most of the Miskitus Cay coral reef system is < 30 m deep, with high wave energy and low visibility that creates an environment dominated by seagrasses, octocoral gardens, coral reef patches, and reef crests built mainly by *Acropora* spp. and *Millepora* spp. Both coral species are known as the main reef crest builders in the Caribbean (Kramer and Kramer 2000). This high species richness and great cover of *A. palmata*, *A. prolifera* and *A. cervicornis* in some reefs of Miskitus Cays suggests a quality habitat complex unknown

TABLE 6. *General characterization of coral colonies in August 2001 by site. sd = standard deviation.*

Site	Diameter, cm ($\bar{x} \pm sd$)	Diameter, cm ($\bar{x} \pm sd$)	Recent mortality (%) ± sd	Old mortality (%) ± sd	Total mortality (%) ± sd	Ratio Live:Dead	% Bleached	% Disease
Macandra	59.4 ± 37.8	49.4 ± 31.4	0.0	24.6 ± 29.2	24.6 ± 29.2	3.1	0.0	0.0
Maxide	93.3 ± 87.0	65.6 ± 59.6	0.0	26.1 ± 36.5	26.1 ± 36.5	2.8	0.0	0.0
Dump	62.5 ± 39.6	52.9 ± 30.6	0.0	34.3 ± 24.7	34.3 ± 24.7	7.3	0.0	0.0
Bojotle	54.4 ± 30.2	35.6 ± 19.7	1.0 ± 1.8	49.4 ± 33.3	50.4 ± 35.1	1.0	12.5	0.0
Ahuya	40.0 ± 23.2	23.4 ± 34.6	8.0 ± 17.9	8.0 ± 12.5	16.0 ± 30.4	5.3	20.0	0.0
Creole	49.8 ± 25.4	43.1 ± 23.4	12.6 ± 24.9	16.6 ± 25.5	29.2 ± 50.4	2.4	19.0	0.0
Miskitu	63.3 ± 44.5	35.0 ± 28.3	0.0	13.3 ± 30.3	13.3 ± 30.3	6.5	0.0	0.0
London	30.0 ± 0.0	50.0 ± 0.0	0.0	25.0 ± 0.0	25.0 ± 0.0	3.0	0.0	0.0
Witties	45.0 ± 24.4	33.4	24.1	20.5 ± 30.6	44.6 ± 32.1	3.7	9.1	4.5
Franklin	55.9 ± 26.8	42.5 ± 27.3	0.0	31.8 ± 27.5	31.8 ± 27.5	2.1	0.0	0.0
Ned Thomas	60.8 ± 36.8	51.7 ± 31.4	0.1 ± 0.4	49.5 ± 31.1	49.6 ± 31.5	1.0	3.8	0.0
Lamarcka	67.7 ± 36.7	57.5 ± 37.4	4.5 ± 16.6	20.8 ± 23.9	25.3 ± 40.5	2.9	0.0	4.6
Nasa	67.8 ± 30.5	51.6 ± 29.1	0.0	54.2 ± 37.8	54.2 ± 37.8	0.8	0.0	0.0
Wiplyn	54.0 ± 27.9	38.0 ± 13.0	4.0 ± 6.5	7.0 ± 11.0	11.0 ± 17.5	8.1	0.0	40.0
Toro	75.2 ± 50.6	52.6 ± 25.3	2.4 ± 5.8	23.9 ± 26.9	26.3 ± 32.8	2.8	14.3	4.8
Hamkira	46.0 ± 24.3	36.7 ± 18.3	0.0	8.3 ± 19.4	8.3 ± 19.4	11.1	0.0	0.0
Yanka Laya	84.2 ± 36.3	69.5 ± 34.4	0.0	20.8 ± 23.6	20.8 ± 23.6	3.8	0.0	0.0
Ryan	66.0 ± 35.3	42.0 ± 25.8	0.0	29.2 ± 33.5	29.2 ± 33.5	2.4	0.0	0.0
$\bar{x}$, all Cayos Miskitus	**59.7**	**46.2**	**3.2**	**25.7**	**28.9**	**3.9**	**4.0**	**3.0**

in much of the Caribbean and is very important in terms of conservation. These very fragile species have been intensively destroyed in other reefs of the Caribbean by hurricanes, anchoring and diving (Precht et al. 2002).

Miskitus Cays has high coral richness (39 species) similar to other sites in the Caribbean (reviewed in Fonseca 2003, Ryan and Zapata 2003). The reef crest complexity index (1.84) found at depths < 5 m is also high, suggesting high reef complexity. Recent and total mortality in Miskitus Cays was lower (3.2 and 27.6%, respectively) than in other shallow reefs from the Mesoamerican Reef System (18 and 49%, respectively); however, mean bleaching and disease incidences were similar (Kramer and Kramer 2000). Several diseases known to be widespread throughout the wider Caribbean (Green and Bruckner 2000) were also found in Miskitus Cays, and the BBD showed the highest frequency (Kramer and Kramer 2000). Dark Spot Disease was most common on massive *Siderastrea* and *Montastraea* (Bruckner 2001), and the WBD was found in *Acropora* as expected. White Band Disease is thought to be a major factor in the decline of elkhorn and staghorn corals in the wider Caribbean (Aronson and Pre-

TABLE 7. *Mean diameter (cm) of dominant coral colonies in August 2001 by site.*

Site	*Siderastrea siderea*	*Montastraea annularis*	*Montastraea faveolata*	*Diploria strigosa*	*Acropora palmata*
Macandra	38.3	–	116.7	50.0	58.3
Maxide	–	–	–	27.5	88.0
Dump	30.0	–	51.4	30.0	75.4
Bojotle	38.8	–	76.7	50.0	–
Ahuya	25.0	40.0	52.5	–	–
Creole	37.5	45.0	73.6	–	–
Miskitu	25.0	–	–	–	103.3
London	–	–	–	30.0	–
Witties	–	–	–	35.0	75.0
Franklin	48.3	–	–	56.0	78.1
Ned Thomas	40.0	–	70.0	–	84.1
Lamarcka	13.6	–	45.0	25.0	83.8
Nasa	30.0	–	74.5	25.0	80.0
Wiplyn	60.0	–	–	40.0	–
Toro	–	–	55.0	–	62.5
Hamkira	40.0	–	55.0	26.7	98.8
Yanka Laya	–	–	60.0	–	92.3
Ryan	55.0	–	110.0	–	50.0

cht 2001), causing major changes in the composition and structure of reefs (Green and Bruckner 2000).

The highest incidence of disease was found in Wiplyn, and may be related to its proximity to MARAS, the largest human settlement in the area where people are discharging waste water and solids directly into the sea. The incidence and prevalence of diseases may also increase when corals are stressed by sedimentation, hurricanes, nutrients, toxic chemicals and warmer-than-normal temperatures (Richardson 1998), as is evidenced by the imports from Hurricane Mitch in 1998 (Kramer and Kramer 2000), and other earlier events (Jameson 1996). Jameson (1996) did not find corals with active diseases, although he observed low levels of bleaching and anchor damage. In this study, no anchor damage was found because the highest reef development occurs in the crests and boats do not anchor directly over them. However, even though large coral colonies within reef crests are dispersed it would be better if mooring buoys were installed in diving and fishing sites to protect the coral colonies. Finally, in 1995 and 1998, late summer temperature increased from 29.5 to 31.1°C throughout the Caribbean, and this coincided with several bleaching reports (Guzmán and Guevara 1998, McField 1999). Apparently, shallow reefs experienced catastrophic losses due to the initial bleaching but now show minimal signs of remnant bleaching (Kramer and Kramer 2000). The coral reef in the Miskitus Cays appear to have recovered because observed mortality was low.

Most of the Miskitus reef system is in good health, especially in the south and southwest sections. The sites in best condition within the Miskitus Cays Biological Reserve are Nasa reef in the southwest, the protected side of the fringing reef around Hamkira and Yanka Laya in the northwest, and to a lesser extent Lamarcka in the southwest, Franklin reef and Ned Thomas in the south, and Creole Bar-Blue Channel in the North. Blue Channel is a potential site for reef fishes' reproductive aggregations. The most deteriorated sites, with non-coralline algae overgrowth and diseases, are those in the west-northwest, especially Ryan, North West reef, Toro Cay, Gigi and Glena Bar and some reef patches around Hamkira and Sammy. This appears to be due to sediments, and associated nutrients and pesticides coming from the deforested mangroves of MARAS and the runoff of Coco River.

The mean live coral cover for Miskitus Cays is relatively high (43%) compared to other continental Caribbean coral reefs (Table 9), with 7 sites having higher live coral (>50%) than algae cover. However, most sites had lower coral cover than algae cover. The percentage of non-coralline algae was higher than the percentage of coralline algae (Table 9) as is typical of the Caribbean region due to a combination of hurricanes, bleaching, diseases, eutrophication, and low herbivory rates as a consequence of over fishing and *Diadema* mass mortality (Goreau et al. 1998, Hayes and Goreau 1998, Kramer and Kramer 2000). The most diverse and structurally complex coral reefs reported for the Central American coast were Belize, Honduras and Panama (Cortés 1997). However, Miskitus Cays has a great development of

TABLE 8. *Relative coverage of benthic substrate (n = 3; "-" = no data) by site in August 2001.*

Site	% Live coral	% Dead coral with algae	% Non-coralline algae	% Coralline algae	% Total algae	Sediment	Others	Difference between % live coral and % algae
Macandra	46.3 ± 13.6	17.9 ± 3.7	21.9 ± 17.1	11.4 ± 1.4	51.2 ± 13.6	0.8 ± 1.4	1.6 ± 1.4	-4.9
Maxide	32.5 ± 19.9	25.2 ± 30.1	36.6 ± 47.5	4.9 ± 8.4	66.7 ± 20.7	0.8 ± 1.4	-	-34.2
Dump	64.2 ± 5.1	27.6 ± 15.7	4.1 ± 7.0	0.8 ± 1.4	32.5 ± 7.4	-	3.2 ± 3.7	31.7
Bojotle Kira	13.0 ± 11.0	-	84.6 ± 7.8	-	84.6 ±7.8	-	2.4 ± 4.2	-71.6
Ahuya Luphia	22.8 ± 5.1	2.4 ±0.0	43.9 ± 10.6	19.5 ± 6.4	65.8 ± 8.8	6.5 ± 9.2	4.9 ± 4.9	-43
Creole Bar	42.3 ± 2.8	4.9 ± 4.2	43.1 ± 9.8	7.3 ± 2.4	55.3 ± 3.7	-	2.4 ± 2.4	-13
Miskitu reef	33.3 ± 1.4	66.7 ± 1.4	-	-	66.7 ± 1.4	-	-	-33.4
London reef	44.7 ± 27.0	55.3 ± 27.0	-	-	55.3 ± 27.0	-	-	-10.6
Witties	33.3 ± 7.4	66.7 ± 7.4	-	-	66.7 ± 7.4	-	-	-33.4
Franklin reef	49.6 ± 29.5	41.5 ± 28.8	4.1 ± 7.0	1.6 ± 2.8	47.2 ± 28.3	1.6 ± 2.8	1.6 ± 2.8	2.4
Ned Thomas	62.6 ± 16.2	30.1 ± 11.0	4.9 ± 4.9	1.6 ± 2.8	36.6 ± 16.0	-	0.8 ± 1.4	26
Lamarcka	56.9 ± 17.1	35.0 ± 23.0	4.9 ± 8.4	1.6 ± 2.8	41.5 ± 17.6	-	1.6 ± 2.8	15.4
Nasa reef	59.4 ± 12.0	39.8 ± 13.4	-	0.8 ± 1.4	40.6 ± 12.0	-	-	18.8
Wiplyn	31.7 ± 10.6	68.3 ± 10.6	-	-	68.3 ± 10.6	-	-	-36.6
Toro cay	43.9 ± 11.2	24.4 ± 8.4	16.3 ± 5.1	14.6 ± 8.8	55.3 ± 11.5	0.8 ± 1.4	-	-11.4
Hamkira	56.9 ± 22.7	6.5 ± 6.1	30.1 ± 16.6	1.6 ± 1.4	38.2 ± 18.0	0.8 ± 1.4	4.1 ± 3.7	18.7
Yanka Laya	58.5 ± 23.3	32.5 ± 34.3	8.9 ± 15.5	-	41.5 ± 23.3	-	-	17
Ryan	29.3 ± 22.4	9.8 ± 9.8	52.8 ± 32.4	-	62.6 ± 24.4	7.3 ± 12.7	0.8 ± 1.4	-33.3
Total $\bar{x}$ for Miskitus Cays (n=54)	**43.4 ± 14.8**	**30.8 ± 26.0**	**19.8 ± 27.0**	**3.7 ± 6.4**	**54.2 ± 14.1**	**2.6 ± 2.9**	**2.3 ± 1.4**	**- 10.8**

TABLE 9. *Comparison of mean live coral and algae coverages between Miskitus Cays and other coral reefs of the Central American Caribbean coast.*

Country	Reef name	Reef type	% Live coral	% Algae	Year	Reference
Panamá	Bocas del Toro	Insular	27	21	1999	CARICOMP 1999
Costa Rica	Cahuita	Continental	13	60	1999	CARICOMP 1999
Nicaragua	Great Corn Island	Insular	36	37	1998	CARICOMP 1999
Nicaragua	Cayos Miskitus	Insular	43	54	2001	This study
Honduras	Roatán	Insular	34	38	1997	Fonseca and Radawsky 1997
Guatemala	Punta Manabique	Continental	9	65	2000	Fonseca 2003
Belize	Carrie Bow Cay	Insular	16	65	1997	CARICOMP 1999
Belize	Calabash Cay	Insular	10	58	1997	CARICOMP 1999
México	Puerto Morelos	Continental	1	93	1999	CARICOMP 1999

reef crests, and Miskitus Cays and Corn Island should be considered high quality reefs suited for a high conservation status in the Caribbean because of their high live coral cover and diversity, and low mortality.

Coral reef degradation in the American region is caused mainly by increased influx of terrigenous sediments (Rogers 1986, Ginsburg 1994) primarily due to deforestation, uncontrolled coastal development, and inappropriate agricultural practices (Cortés and Risk 1985). While suspended sediment loads appear to be the greatest human threat to Nicaragua´s reefs (Ryan et al. 1998), unregulated fishing activities have also caused damage. Despite these multiple threats, Nicaragua still lacks a concrete management strategy for its coastal and marine resources due mainly to a general lack of political awareness about the key role that coral reefs play in supporting fisheries and biodiversity, institutional and human capacity gaps and inadequate legislation for reef conservation (Ryan and Zapata 2003). Integrated management of Nicaraguan river basins and coastal-marine reserves is urgent and this can be promoted by extending land-conservation approaches to marine ecosystem biodiversity.

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Literature Cited

Alevizon, W. 1993. A preliminary survey of the coral reefs and reef fisheries of the Miskitu Cays, Nicaragua. Technical Report, Caribbean Conservation Corporation, Miami, Florida, USA, 16 p.

Aronson, R.B. and W.F. Precht. 2001. Evolutionary palaeoecology of Caribbean coral reefs. In: Allmon, W.D. and D.J. Bottjer, eds. Evolutionary Paleoecology: The Ecological Context of Macroevolutionary Change. Columbia University Press, New York, NY, USA, p. 171-233.

Bruckner, A.W. 2001. Coral health and mortality: Recognizing signs of coral diseases and predators. In: Humann, P. Reef Coral Identification. Florida, Caribbean, Bahamas. New World Publications, Inc. Jacksonville, FL, USA, p. 240-271.

Cairns, S.D. 1982. Stony corals (Cnidaria: Hydrozoa, Scleractinia) of Carrie Bow Cay, Belize. In Rützler, K. and I.G. Macintyre, eds. The Atlantic Barrier Reef Ecosystem at Carrie Bow Cay, Belize, 1: Structure and Communities. Smithsonian Contributions to the Marine Sciences 12:271-301.

CARICOMP. 1999. Caribbean Coastal and Marine Productivity (CARICOMP) (database). Jamaica. http://www.ccdc.org.jm/caricomp.html

Cortés, J. and M. Risk. 1985. A reef under siltation stress: Cahuita, Costa Rica. Bulletin of Marine Science 36:339-356.

Cortés, J. 1996-1997. Biodiversidad marina de Costa Rica: Filo Cnidaria. Revista de Biología Tropical 44/45:323-334.

Cortés, J. 1997. Status of the Caribbean coral reefs of Central America. Proceedings, 8th International Coral Reefs Symposium 1:335-340.

Cortés, J. 2003. Latin American coral reefs. Elsevier Science B.V., Amsterdam, The Netherlands, 497 p.

Fonseca, E. A. 2003. Coral reefs of Guatemala. In: J. Cortés, ed. Latin American Coral Reefs. Elsevier Science B.V., Amsterdam, The Netherlands, p. 159-169.

Fonseca, E. A. and J. Radawsky. 1997. Evaluación rápida del arrecife coralino, los límites y zonificación de la Reserva Marina Sandy Bay/West End, Roatán, Honduras. Prioridades de manejo y recomendaciones para el Plan de Monitoreo a largo plazo. Technical Report, Wildlife Conservation Society, FL, USA, 31 p.

Ginsburg, R.N. 1994. Proceedings, Colloquium on Global Aspects of Coral Reefs, Health, Hazards and History, University of Miami, Miami, FL, USA, June 10-11, 1993. p. 240-246.

Goreau, T. J., J. Cervino, M. Goreau, R. Hayes, M. Hayes, L. Richardson, G. Smith, K. DeMeyer, I. Nagelkerken, J. Garzon-Ferrera, D. Gil, G. Garrison, E. H. Williams, L. Bunkley-Williams, C. Quirolo, K. Patterson, J. Porter, and K. Porter. 1998. Rapid spread of Caribbean coral reef diseases. Revista de Biología Tropical 46:157-171.

Green, E. and A.W. Bruckner. 2000. The significance of coral disease epizootiology for coral reef conservation. Biological Conservation 96:347-361.

Guzmán, H.M. 1998. Diversity of stony, soft and black corals (Anthozoa: Scleractinia, Gorgonacea, Antipatharia; Hydrozoa: Milleporina) at Cayos Cochinos, Bay Islands, Honduras. Revista de Biología Tropical 46:75-80.

Guzmán, H. M. and C. A. Guevara. 1998. Arrecifes coralinos de Bocas del Toro, Panamá: I. Distribución, estructura y estado de conservación de los arrecifes continentales de la Laguna de Chiriquí y la Bahía Almirante. Revista de Biología Tropical 46:601-622.

Hallock, P., A.C. Hine, G. A. Vargo, J. A. Elrod, and W. C. Jaap. 1988. Platforms of the Nicaraguan Rise: examples of the sensitivity of carbonate sedimentation to excess trophic resources. Geology 16:1104-1107.

Harrington, G.A. and V. Gallucci. 1996. Analysis of the artisanal fisheries and the potential for co-management in the Miskito Cays Protected Area, Atlantic coast of Nicaragua. Environmental Initiative of the Americas fisheries project. Technical Report, United States Agency for International Development, Washington, D.C., USA, 65 p.

Hayes, R.L. and N.I. Goreau. 1998. The significance of emerging diseases in the tropical coral reef ecosystem. Revista de Biología Tropical 46:173-185.

Humann, P. 1993. Reef Coral Identification. Florida, Caribbean, Bahamas. New World Publications, Inc. Jacksonville, FL, USA, 239 p.

Jameson, S.C. 1996. Coral reef ecosystem survey and management recommendations. National Oceanic and Atmospheric Administration. Environmental Initiative of the Americas Fisheries Project. Technical Report, United States Agency for International Development, Washington, D.C., USA, 14 p.

Kramer, P.A. and P.R. Kramer. 2000. Ecological status of the Mesoamerican Barrier Reef System. Impacts of Hurricane Mitch and 1998 coral bleaching. Technical Report, Rosenstial School of Marine and Atmospheric Sciences/World Bank. Miami, FL, USA, 73 p.

Lang, J.C. (ed.). 2003. Status of coral reefs in the Western Atlantic. Results of initial surveys, Atlantic and Gulf Rapid Reef Assessment (AGRRA) Program. Atoll Research Bulletin 496:630.

Maradiaga, J. 1998. Proyecto desarrollo integral de la pesca artesanal de la RAAN (DIPARAAN). Componente de validación y transferencia tecnológica. Technical Report. V Crucero Exploratorio, Puerto Cabezas, Nicaragua, 50 p.

Marshall, M. 1994. Seagrasses of the Miskito Cays Biological Reserve. Environmental Initiative of the Americas Fisheries Project. Technical Report, United States Agency for International Development, Washington, D.C., USA, 12 p.

McField, M.D. 1999. Coral response during and after mass bleaching in Belize. Bulletin of Marine Science 64:155-172.

Murray, S.P., S.A. Hsu, H.H. Roberts, E.H. Owens, and R.L. Crout. 1982. Physical processes and sedimentation on a broad, shallow bank. Estuarine, Coastal and Shelf Science 14:135-157.

Nadon, M.-O. and G. Stirling. 2006. Field and simulation analyses of visual methods for sampling coral cover. Coral Reefs 25:177-185.

Ogden, J.C. and E.H. Gladfelter. 1983. Coral reefs, seagrass beds and mangroves: their interaction in coastal zones of the Caribbean. Reports in Marine Science 23. United Nations Educational, Scientific and Cultural organization, Regional Office for Science and Technology for Latin America and the Caribbean. Montevideo, Uruguay, 133 p.

Phillips, R.C., R.L. Vadas, and N. Ogden. 1982. The marine algae and seagrasses of the Miskito Bank, Nicaragua. Aquatic Botany 13:187-195.

Precht, W.F., A.W. Bruckner, R.B. Aronson, and R.J. Bruckner. 2002. Endangered acroporid corals of the Caribbean. Coral Reefs 21:41-42.

Rogers, C. S. 1986. Responses of coral reefs and reef organisms to sedimentation. Marine Ecology Progress Series 62:185-202.

Rogers, C. S., G. Garrison, R. Grober, Z.-M. Hillis, and M.A. Franke. 1994. Coral reef monitoring manual for the Caribbean and Western Atlantic. Technical Report. The Nature Conservancy/World Wildlife Fund, Miami, FL. USA, 89 p.

Richardson, L.L. 1998. Coral diseases: What is really known? Trends in Ecology and Evolution 13:438-443.

Ryan, J. and Y. Zapata. 2003. Nicaragua´s coral reefs: status, health and management strategies. In: J. Cortés, ed. Latin American Coral Reefs. Elsevier Science B.V., Amsterdam, The Netherlands, p. 203-222.

Ryan, J.D., L.J. Miller, Y. Zapata, O. Downs, and R. Chan. 1998. Great Corn Island, Nicaragua. In: Kjerfve, B. CARICOMP. Caribbean Coral Reef, Seagrass and Mangrove Sites. Technical Report, United Nations Educational, Scientific and Cultural organization, Paris, France, p. 95-105.

Siirila, E. 1992. Status and value of the Bay Islands coral reefs, Honduras. Technical report. United Nations Development Project, Honduras, 158 p.

USAID. 1996. Cayos Miskitos. Environmental Initiative of the Americas Fisheries Project. October, 1995 to September, 1996. Technical Report. United States Agency for International Development, Washington, D.C., USA, 51 p.

Valerio, L. 2001. Mapa de hábitats marinos de Cayos Miskitus. MARENA/URACCAN/ PROARCA-COSTAS/TNC. Map. Managua, Nicaragua, 1 p.

Villeda, E., S. Rodríguez, M. Espinal, G. Cruz, G. Paz, and A. Zúñiga. 1997. Reserva Marina Sandy Bay-West End, Roatán, Islas de la Bahía. Plan de manejo y desarrollo. Versión preliminar. ECO-AMBIENTE, 200 p.

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Habitat Condition and Associated Macrofauna Reflect Differences Between Protected and Exposed Seagrass Landscapes

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HABITAT CONDITION AND ASSOCIATED MACROFAUNA REFLECT DIFFERENCES BETWEEN PROTECTED AND EXPOSED SEAGRASS LANDSCAPES

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ABSTRACT: Seagrass landscape configurations associated with different physical settings can affect habitat-structure and plant-animal relationships. We compared shoal grass *(Halodule wrightii)* habitat and macrofaunal variables between two fragmented seagrass landscapes at barrier-island locations subject to different disturbance regimes. Five seagrass habitat variables including above ground biomass (AGB), shoot number, per shoot biomass, epiphyte biomass and below ground biomass (BGB), differed significantly between the island landscapes. Per shoot biomass and epiphyte biomass also varied significantly over the seagrass growing season; and epiphyte biomass showed a strong landscape-time interaction. Abundances of microgastropods normalized to AGB differed significantly between landscapes. An inverse relationship between the abundance of microgastropods and epiphyte loading suggests a possible functional link. However, additional temporal mismatch between epiphyte loading and microgastropod abundance indicates that controls on epiphyte loading were complex. Seagrass habitat was more fragmented within the Cat Island (CI) landscape. Wind direction and strength imply that the CI landscape experienced more physical disturbance than the Horn Island (HI) landscape. This study highlights some potential links involving landscape configuration, habitat structure, and macrofaunal associations which can be further addressed using hypothesis-driven research.

Introduction

Seagrass ecosystems exist as hierarchically organized habitats in various states of fragmentation, mediated by landscape-scale forces (Pittman et al. 2004). Hierarchical spatial patterns arise from the interaction of broad-scale external effects on habitat configuration and local internal effects on habitat structure (Boström et al. 2006). For example, physical disturbance induces variability in the spatial configuration of patches of varying sizes and interpatch distances within the seagrass landscape (Fonseca and Bell 1998). Furthermore, processes occurring at broad spatial scales may constrain those occurring at local spatial scales (Allen and Starr 1988). Consequently, landscape-scale features, such as areal cover, patch size, and interpatch distance, may covary with habitat-structure (Boström et al. 2006), as expressed by shoot density, above ground biomass (AGB), below ground biomass (BGB), epiphyte loading (Moore and Fairweather 2006) or associated macrofauna (Hovel et al. 2002).

Although macrofaunal associations change with the spatial arrangement of seagrass habitat (Turner et al. 1999, Frost et. al 1999), responses by individual taxa can vary relative to landscape configuration (e.g., patch size and distance) (Bell et al. 2001). The apparent inconsistency reflects the fact that macrofaunal taxa relate individualistically to different environmental scales (Boström et al. 2006), thus accounting for different response thresholds to habitat fragmentation. Seagrass ecosystems also form complex trophic networks defined by internal feedbacks on habitat function, including those exerted by macrofauna (Connolly and Hindell 2006). For example, some bivalves enhance seagrass condition by locally increasing both light accessibility and sediment nutrients (Peterson and Heck 2001). Such links also may be decoupled by broad-scale physical disturbance or habitat fragmentation. Again, critical thresholds in functional links with decreasing habitat connectivity depend on the species' biology and the physical setting (With and Crist 1995, Fonseca and Bell 1998, Monkonnen and Reunanen 1999).

The first step towards understanding habitat function relative to landscape-scale factors is to identify potential habitat-scaling relationships. So we compared shoal grass (*Halodule wrightii)* habitat and macrofaunal metrics during the seagrass growth phase between two barrier- island landscapes exposed to different levels of disturbance. Habitat metrics included: above ground biomass (AGB); epiphyte biomass; shoot number; per shoot biomass; and below ground biomass (BGB); macrofaunal metrics included abundances of microgastropods, peracarid grazers, capitellid polychaetes, *Neanthes* polychaetes, and macrofaunal diversity. Our working hypothesis was that seagrass landscape, habitat and faunal metrics should differ concertedly between more disturbed Cat Island (CI) and less disturbed Horn Island (HI) landscapes.

Study Area

Two seagrass landscapes separated by 45 km extended along the north-central HI shoreline and around the western tip of CI (Figure 1). Horn Island is part of the Gulf Islands National Seashore under the jurisdiction of the US National Park Service. Waters surrounding CI were managed only by state and federal dredge and fill regulations prior to and during the time frame of this study (CI was acquired by US NPS in 2003). The HI landscape: (1) is apparently less exposed to physical disturbance than CI; and (2) has been protected from trawling within 1.6 km of shore since May 1995 by the U.S. National Park Service.

Materials and Methods

Disturbance and habitat fragmentation

Physical disturbance within the CI and HI landscapes over four months prior to and during the study period from 15 May until 9 August 1998 was estimated from hourly measurements of wind direction, wind speed, and wave height taken at NOAA Data Buoy 42007 located off the north point of the Chandeleur Islands (30°05'24" N; 88°46'12" W), 19 km south of HI and 40 km southeast of CI. Monthly mean (± 1 se) wind directions were calculated using circular statistics (Oriana Ver 1.0; Kovach 1994). Salinity was compared between the eastern and western portions of Mississippi Sound using data obtained from the MS Department of Marine Resources for roughly 40 stations during the May – June (80 vs. 90 observations) and July – August (171 vs. 29 observations) periods.

Seagrass fragmentation was quantified from 4m resolution digital aerial photographs of seagrass cover taken in March 1998. ArcGIS 8.2 was used to digitize seagrass patches occurring within 13 hectares both off the west tip of CI and along the northwest central side of HI. The digitized areas coincided with the landscape areas used for this study (Figure 1). Four, one hectare quadrats were randomly placed within each of the two island landscapes with the restrictions that they could not overlap with each other or fall outside of the seagrass-depth contour within designated areas.

Field sampling

Three sites separated by ~ 0.3 km were located within each of the two island landscapes (Figure 1). Three monthly sampling events during the seagrass growth phase ensued on 3 June, 22 June, and 5 August, 1998. At each site, three cores of seagrass and associated macrofauna (i.e., subsamples) were randomly taken within a ~ 0.01 km^2 area using a 16.0 cm diameter plexiglass corer to extract 0.02 m^2

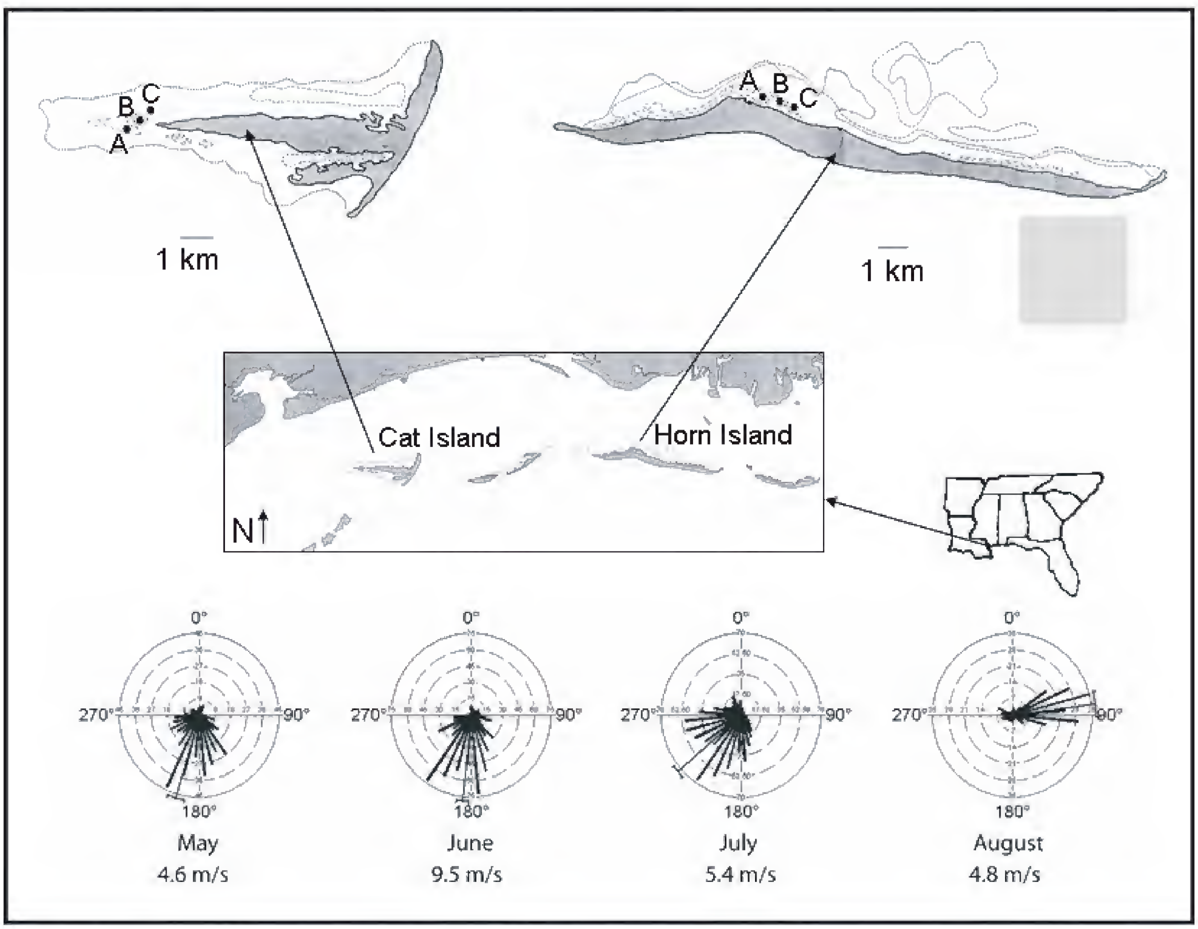

Figure 1.
Map of the study region showing the two barrier-island landscapes and the six sites. Circular graphs depict monthly wind direction vectors, along with mean (± 95% CI) wind velocities measured at NOAA Data Buoy 24007 during the study.

TABLE 1. *Variation in wind and wave strength over the study period between 15 May and 9 August 1998 measured at NOAA Data Buoy 42007 (30°05'24" N; 88°46'12" W). Values represent monthly means (± 1 se) obtained by aggregating hourly data for each day and daily values for each month. Monthly means (± 1 se) of wind directions were calculated using circular statistics, Oriana Ver 1.0 (Kovach 1994). Significant wave height (meters) was calculated as the mean of the highest one third of all of the wave heights during the 20 minute sampling period.*

Month	Wind speed (m sec^{-1})	Wind direction (°)	Significant wave height (m)	Mean wave period (sec)
May 15 - 31	4.56 ± 0.281	196.2 ± 3.11	0.35 ± 0.057	3.00 ± 0.036
June 1 - 30	5.24 ± 0.325	186.0 ± 2.21	0.48 ± 0.033	3.84 ± 0.103
July 1 - 31	4.41 ± 0.249	229.3 ± 2.36	0.27 ± 0.025	2.98 ± 0.237
August 1 - 9	4.79 ± 0.503	82.6 ± 4.31	0.57 ± 0.079	3.71 ± 0.193

sediment plugs to a depth of 15 cm. A total of 54 cores were taken (i.e., 2 landscape areas x 3 sites x 3 cores x 3 periods). A 0.5 mm mesh polypropylene sieve was used to remove fines, while still retaining all plant material and associated macrofauna. Water column salinity (psu), turbidity (NTU), water temperature (°C), water depth (m), substrate type, and photosynthetically active radiation (PAR) (μmol photons m^{-2} sec^{-1}) were recorded for each sampling event.

Laboratory processing

Plant material was carefully separated by gentle rinsing in tapwater and frozen for later processing. Associated coarse sediment and macrofauna were rewashed in a 0.5 mm mesh sieve and preserved in 10% formalin. Ten randomly selected shoots were used to quantify the epiphyte load by scraping shoots and leaves with a dull razor blade. Shoot and epiphyte fractions were dried at 105 °C for 24 h or until a constant weight was obtained, and then weighed to the nearest 0.001 g using an O'Haus microbalance. In addition, remaining AGB and separated BGB fractions were dried and weighed (mg) as described above. Preserved macrofaunal organisms were sorted, identified to the lowest practical taxonomic level, and enumerated.

Data analysis

Metrics for comparing seagrass fragmentation included number of patches, total patch area, patch area percent cover, mean patch size, and standard deviation in patch size. Patch metrics were compared between HI and CI using Students independent-sample t-tests ($p < 0.05$). Two-tailed t-tests were based on assumptions of equal or unequal variance, depending on the outcome of Levene's tests of homogeneity of variance in SPSS 13.0 (SPSS 2004).

Habitat and macrofaunal variables examined included the number of short shoots (shoot number), above-ground biomass without epiphytes (AGB), epiphyte biomass (= arcsine square-root (epiphyte biomass/(epiphyte biomass + AGB))), below ground biomass (BGB), per shoot biomass, microgastropod abundance (normalized to AGB), peracarid grazer abundance (normalized to AGB), capitellid abundance, *Neanthes* abundance, and macrofaunal diversity (Shanon-Wiener H'; base 2). Macrofaunal abundances were log transformed (i..e, $\log_{10}$ (N+1)) prior to analysis.

To accommodate spatial and repeated time effects, the Linear Mixed Models (LMM) procedure was employed in SPSS 13 (SPSS 2004). LMM is very flexible in that it can model covariance and heterogeneous variability in the context of concurrent fixed and random effects (Verbeke and Molenberghs 2000). The Unstructured Covariance Model was fit as it provides the least restricted covariance structure and is equivalent to the multivariate form of Repeated Measures ANOVA. Site was treated as a subject variable and time as a repeated effect. Landscape and time were considered fixed main effects, and the landscape-time interaction term was also included. Tests of fixed effects utilized Type III sums of squares. Cases for LMM comprised means of the three cores per site-time event.

For selected seagrass and macrofaunal variables, means (± 1 se) were plotted for each of the three sites from each island landscape across the three sampling dates. An inverse hyperbolic curve of the form $Y = a X^{-b}$ between the abundance of microgastropods (i.e., *Bittiolum varium* and *Astyris lunata*) and epiphyte mass (as the proportion of total AGB) was fit for the 54 cores.

Results

Disturbance and habitat fragmentation

Wind direction and strength measured at NOAA Data Buoy 42007 implied that the CI landscape was more exposed to physical disturbance than the HI landscape just prior to and during the study period. Winds typically originated from the south-southwest for the three month period between 15 May and 9 August (Figure 1; Table 1). However, CI sites were relatively protected near the end of the study period in early August, when winds primarily originated from the east. Wave action in concert with wind strength was relatively high in June, when the mean wave height was 0.48 ± 0.033 m (mean ± 1 se) while the mean wind velocity was 5.24 ± 0.325 m sec^{-1}. Although HI sites are lo-

cated farther than CI sites from several major freshwater discharge sources, including the Bonnet Carré Spillway, and the Jourdan and Pearl Rivers (90 km vs. 46 km, 67 km vs. 25 km, and 80 km vs. 35 km, respectively), salinity was similar between the two island landscapes during the study period. Salinity averaged 14.8 ± 4.5 ($\bar{x}$ ± 1 sd) vs. 17.0 ± 4.3 between western and eastern portions of Mississippi Sound during May – June 1998; and 22.7 ± 4.5 psu vs. 22.5 ± 4.4 psu during July – August 1998. Other conditions including water temperature, depth, turbidity, sediment composition (i.e., sand) and light were also similar between areas.

Seagrass habitat was notably more fragmented within the CI landscape than in the HI landscape (Table 2). The number of patches, total patch area, and mean patch size, were significantly different between island landscapes (t-tests; all $p \leq 0.002$). The mean number of 20.75 patches per hectare at CI was more than three-fold higher than at HI; whereas, the mean total patch area of 4520.89 m^2 per hectare (i.e., 45.21% seagrass cover) at HI was nearly four-fold higher than at CI (i.e., 12.35% seagrass cover). The grand mean patch size of 645.84 m^2 at HI was ten-fold larger than at CI.

Seagrass habitat variables

All five seagrass habitat variables including AGB, shoot number, per shoot biomass, epiphyte biomass, and BGB differed significantly between island landscapes (Table 3). Per shoot biomass and epiphyte biomass, also varied significantly in time. The landscape-time interaction was significant for epiphyte biomass, and marginally significant for BGB.

AGB was usually higher within the HI landscape, especially in August (Figure 2A). Over the study period, mean AGB ranged from 0.59 to 0.96 g dw per 0.02 m^2 at HI; whereas it ranged from 0.45 to 0.62 g dw per 0.02 m^2 at CI. Conversely, shoot number was slightly higher at CI (Figure 3A). However, per shoot biomass was clearly higher at HI than at CI, and also increased during the study period (Figure 3B). Between June and August, per shoot biomass increased from 0.0061 to 0.0113 g dw at HI, whereas it increased from 0.0040 to 0.0067 g dw at CI. Epiphyte biomass was markedly higher at the CI landscape (Figure 4A); however, this metric also declined markedly in time at CI, while remaining nearly the same at HI. Monthly epiphyte biomass ranged from 3.7 to 9.4 percent of total AGB at HI; whereas it ranged much higher, from 11.3 to 41.3 percent of total AGB at CI. BGB values were also consistently higher at HI over the three sample periods (Figure 2B); although BGB increased over time at CI (Table 3). Mean BGB ranged from 3.0 to 3.9 g dw per 0.02 m^2 at HI; whereas BGB ranged from 0.9 to 1.6 g dw per 0.02 m^2 at CI.

Macrofaunal variables

Macrofaunal species richness (S) was similar between the two barrier island landscapes: 86 taxa were collected from both of the landscapes, each of which yielded 32 unique taxa. Thus, a total of 118 taxa were enumerated over the study period. Diversity (i.e., macrofaunal H'; base 2) was the only macrofaunal metric for which the landscape-time interaction was even marginally significant. Otherwise, Diversity was similar between landscapes and sample periods; mean diversity ranged from

TABLE 2. *Seagrass landscape metrics reflecting differences in habitat fragmentation from digital aerial images of the Cat Island and Horn Island landscapes taken in March 1998. Values represent means (± 1 sd) of metrics for four randomly selected 1-hectare (ha) quadrats within each designated 13.2-ha bounding plot area.*

CAT ISLAND

Plot	Total Plot Area (m^2)	Number of Patches	Total Patch Area (m^2)	% Seagrass Cover	$\bar{x}$ Patch Size (m^2)	Patch Size sd (m^2)
1	10000	17	1316.14	13.16	77.42	103.67
2	10000	23	617.59	6.18	26.85	78.91
3	10000	21	2439.89	24.40	116.19	115.24
4	10000	22	567.42	5.67	25.80	24.66
Aggregate	**40000**	**83**	**4941.04**	**12.35**	**59.53**	**84.59**

HORN ISLAND

Plot	Total Plot Area (m^2)	Number of Patches	Total Patch Area (m^2)	% Seagrass Cover	$\bar{x}$ Patch Size (m^2)	Patch Size sd (m^2)
1	10000	5	4558.46	45.58	759.74	1559.66
2	10000	7	5439.96	54.40	777.14	912.15
3	10000	8	3300.78	33.01	412.60	876.66
4	10000	7	4784.37	47.84	683.48	905.43
Aggregate	**40000**	**27**	**18083.55**	**45.21**	**645.84**	**1020.11**

TABLE 3. *Linear Mixed Models (LMM) results for nine seagrass habitat and macrofaunal variables. Landscape and time are considered fixed effects. Time is also considered to be a repeated factor and sites are regarded as subjects. Unstructured (= completely general covariance matrix) LMM model used, as explained in the text; No. model parameters = 12. AGB = Above Ground Biomass; BGB = Below Ground Biomass; GRZ PERACARIDS = Grazing Peracarids. Faunal abundances tested on log_{10} (N+1) scale. Epiphyte biomass tested as arcsine square-root proportion of total AGB. Bold = significant, Bold underline = marginally significant.*

Dependent Variable	-2RLL	LNDSCP F	p	TIME F	p	LND x TIME F	p
AGB	-14.546	8.666	**0.042**	0.973	0.453	2.027	0.247
SHOOT NUMBER	102.216	12.597	**0.024**	0.345	0.727	0.430	0.677
PER SHOOT BIOMASS	-72.863	118.749	**<0.001**	15.783	**0.013**	2.577	0.191
EPIPHYTE BIOMASS	-39.295	123.619	**<0.001**	47.888	**0.002**	31.608	**0.004**
BGB	25.592	17.508	**0.014**	0.257	0.786	4.464	**<u>0.096</u>**
MICROGASTROPODS	-14.664	155.083	**<0.001**	27.161	**0.005**	0.413	0.687
GRZ PERACARIDS	3.574	0.602	0.481	2.053	0.243	1.140	0.406
CAPITELLIDS	-1.550	6.557	**<u>0.063</u>**	2.971	0.162	3.658	0.125
NEANTHES	-8.639	0.421	0.552	1.961	0.255	1.992	0.251
DIVERSITY	7.336	0.906	0.395	1.640	0.302	5.488	**<u>0.071</u>**

2.32 to 2.82 per 0.06 m^2 (i.e., for 3 cores combined).

The overall macrofaunal density was notably four-fold higher at HI (mean (± 1 se) = 247.1 (± 35.7) per 0.02 m^2) than at CI (mean (± 1 se) = 61.0 (± 5.0) per 0.02 m^2). Typical seagrass-associated macrofauna included the amphipods, *Ampelisca holmesi* and *Cymadusa compta*; the isopods, *Edotea triloba* and *Erichsonella attenuata*; the gastropods, *Astyris lunata* and *Diastoma varium*, the caridean shrimp, *Hippolyte zostericola*, *Latreutes parvulus*, and *Palaemonetes pugio*; the brachyuran crab, *Callinectes sapidus*; and anomuran crabs, *Pagurus* spp. The macrofauna primarily comprised microgastropods (52.0% HI vs. 10.4% CI), peracarid crustaceans (18.1% HI vs. 9.6% CI), and infaunal polychaetes (13.6% HI vs. 23.4% CI).

Microgastropod abundances differed significantly between landscapes; abundances were higher by an order of magnitude at HI (Figure 4B). They also varied significantly in time. Microgastropods comprised 84% *Bittiolum varium* and 16% *Astyris lunata*. Changes in log abundances were parallel across the sample period between landscapes, first decreasing, and then increasing to the highest levels. Mean microgastropod abundances ranged from 33.0 to 296.4 per g dw AGB at HI; and from 5.3 to 22.7 per g dw AGB at CI. A significant inverse hyperbolic relationship was apparent between the abundance of microgastropods and the epiphyte load (Figure 5). Although low epiphyte values corresponded with a fairly wide range in microgastropod abundance, high epiphyte values (i.e., ≥ 20 percent of total AGB) never occurred in association with high snail abundances.

When scaled to AGB, abundances of peracarid grazers did not differ significantly between island landscapes; mean abundances ranged from 8.4 to 35.7 per g dw AGB. Major peracarid grazers included the ampithoid amphipods (*Ampithoe* and *Cymadusa*) and the isopod, *Erichsonella*. Infaunal polychaetes mostly consisted of the nereid, Neanthes (65% of total) and capitellids (18% of total; mainly *Capitella* and *Mediomastus*). Mean abundances of *Neanthes* ranged widely from 4.3 to 34.9 per 0.02 m^2. Mean abundances of

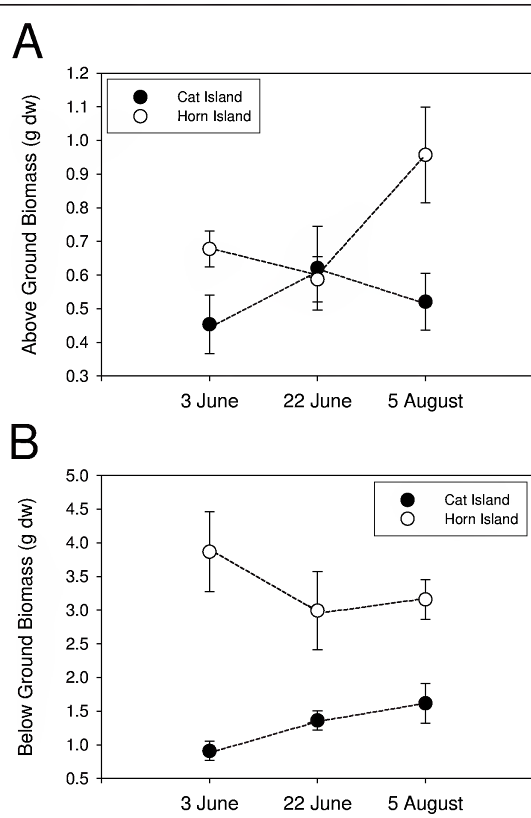

Figure 2.
A. Variation in mean (± 1 se) AGB during the study period.
B. Variation in mean (± 1 se) BGB during the study period.

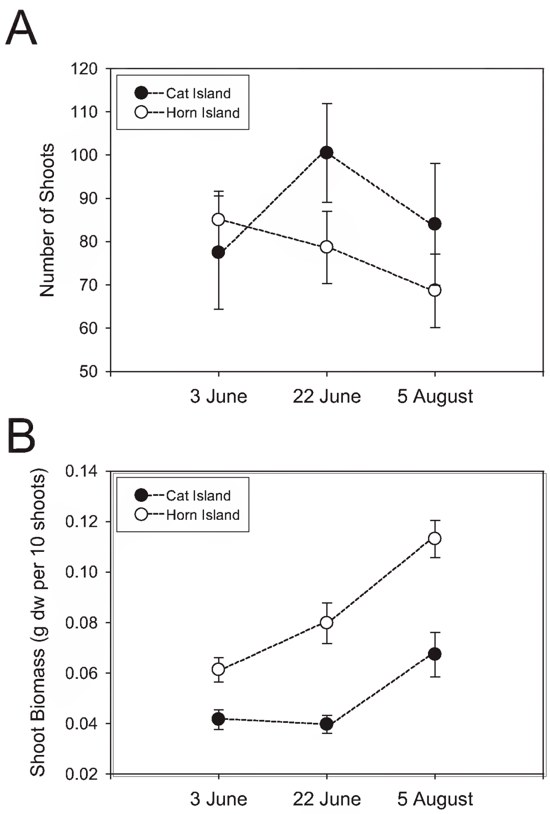

Figure 3.
A. Variation in the mean (± 1 se) number of shoots during the study period.
B. Variation in mean per shoot biomass (± 1 se) (per 10 shoots) during the study period.

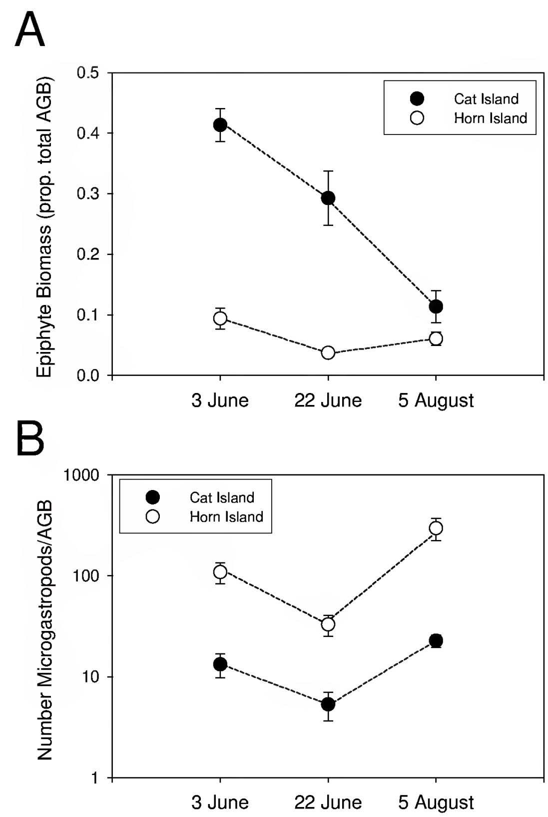

Figure 4.
A. Variation in the mean (± 1 se) epiphyte biomass as a proportion of total ABG during the study period.
B. Variation in the mean (± 1 se) abundance of microgastropod snails scaled to AGB.

capitellids ranged from 1.7 to 11.3 per 0.02 m^2 at HI; whereas at CI they ranged noticeably lower, from 0.8 to 1.4 per 0.02 m^2; and the difference in capitellid abundances between island landscapes was marginally significant (Table 3).

Discussion

Fonseca and Bell (1998) established that physical setting is the main determinant of seagrass landscape configurations, ranging from continuous to widely-dispersed patches with increasing disturbance. Other former studies also document patchy landscapes in high energy environments (Boström et al. 2006). The CI landscape was more exposed to physical disturbance in the form of winds and wave action than HI. Although both seagrass landscapes were fragmented, the seagrass landscape was correspondingly more fragmented at CI (i.e., 12% cover at CI vs. 45% cover at HI). Fonseca and Bell (1998) proposed the critical threshold of ~ 50% coverage, below which the loss of structural habitat integrity accelerates with increasing fragmentation.

Low seagrass coverage at CI corresponded with relatively low per shoot biomass, high epiphyte loading, and low BGB. A feasible link between effects of external and internal processes on habitat function might involve epiphyte loading. High epiphyte loading is known to suppress photosynthetic efficiency by preempting light, water column nutrients, carbon, and oxygen (Sand-Jensen 1977, Sand-Jensen et al. 1985). Epiphyte loading may also exacerbate physical disturbance by increasing hydrodynamic impacts (Jernakoff et al. 1996).

A recent paradigm shift in seagrass ecology recognizes the relative importance of top-down rather than bottom-up controls on epiphyte loading (Jernakoff et al. 1996); and calls for full consideration of the role of plant-animal interactions in studies of eutrophication effects in seagrass ecosystems (Hughes et al. 2004). However, attempts to link landscape-scale metrics and faunal responses in seagrass ecosystems have been equivocal (Bell et al. 2001). Macrofauna potentially exert internal feedbacks on habitat function in a variety of ways (Connolly and Hindell 2006), and these feedbacks might also be susceptible to disruption from physical disturbance and resulting habitat fragmentation.

Any important plant-animal relationship requires two conditions. First, the animal should exhibit either direct or indirect functional links to plant habitat via actions affecting plant condition. Examples include predation on grazers, epiphyte grazing, or nutrient retention or delivery. Second, the strength of the functional link should vary with specific density of the animal (sensu Murphey and Fonseca 1995), or with abundance normalized to some habitat met-

ric (e.g., AGB). The latter condition also implies potential sensitivity to landscape-scale changes in seagrass habitat. Accordingly, we examined abundances of microgastropods and peracarid grazers in relation to ABG. In this study, the clearest indication of a functional plant-animal link was an inverse relationship between the abundance of microgastropods and epiphyte loading. The dominant microgastropod, *Bittiolum varium*, is an important grazing component in seagrass ecosystems (van Montfrans et al. 1982, Edgar 1990, Neckles et al. 1993). Thus, microgastropods potentially enhance seagrass condition by removing epiphytes and redirecting nutrients to the sediments. Recently, Fong et al. (2000) showed that gastropod grazers, *Clithon* spp., directly improved the condition of *Zostera japonica* by removing epiphytic algae. But gastropod densities were positively correlated with seasonally high epiphytic loading in their system.

Extremes in spatial configurations of seagrass habitat must bracket habitat fragmentation thresholds for individual taxa, above which dispersal and recolonization becomes ineffective (Monkkonen and Reunanen 1999, Boström et al. 2006). For example, some threshold level of habitat fragmentation might impair the seagrass-epiphyte-microgastropod relationship by disrupting dispersal (Bell et al. 2001) or by increasing the chance of local extinction (Fahrig 2002). Recruitment of *Bittiolum varium* involves the production of seasonal cohorts via a planktonic larval stage that persists for about three weeks in the water column (Qurban 2000). Planktonic dispersal of larval gastropods implicates landscape fragmentation within the context of source-sink dynamics; it would behoove larvae to settle before they are swept away from suitable habitat. Spatial isolation of seagrass beds from sources of planktonic larvae might occur. Extinction rates of macrofaunal populations might also be increased within fragmented seagrass habitat due to edge-effects (i.e. perimeter:area) that foster increased predation or emigration within smaller beds (Bologna and Heck 1999, Hovel and Lipcius 2001).

Additional temporal mismatch between epiphyte biomass and microgastropod abundance suggests that controls on epiphyte loading were complex. This incongruence could have arisen from changes in the rate of seagrass senescence across the summer period (Nelson 1997, Fong et al. 2000). Higher rates of senescence and resultant lower epiphyte loading may occur as rates of seagrass production increase seasonally with water temperature (Peterson and Heck 2001). Another possible cause of seasonal decline in epiphyte loading at CI could involve exacerbated loss of seagrass shoots with high epiphyte loads due to consequent hydrological disturbance (Jernakoff et al. 1996). Seasonal differences in nutrient availability could also limit the development of epiphytes.

Temporal mismatch between epiphyte biomass and microgastropod abundance could also reflect algal successional patterns, possibly involving interactions with microgastropods. Microgastropod grazers consume mostly diatoms (van Montfrans et al. 1982); however, there were clearly large quantities of filamentous epiphytic algae at the CI site (pers. obsv.). An alternate algal successional pattern might be fostered by lower grazing pressure on the biofilms of surfaces of seagrass shoots. As has been shown for various peracarids (Duffy et al. 2001), selective grazing by microgastropods could favor slower growing early successional epiphytes, perhaps by conditioning seagrass surfaces. Alteration of the algal canopy by grazing might also facilitate colonization by early successional epiphytes (Sommer 1999). For whatever reasons, other studies show that the epiphyte community of disturbed seagrass habitat shifts towards filamentous algae and away from diatoms (Pinckney and Micheli 1998).

Although the importance of the seagrass canopy to secondary production is known, the role of the seagrass root-rhizome mat is not well understood. In this study, capitellids appeared to be more abundant at the HI landscape, where BGB was also consistently higher. This suggests a facultative association for this infaunal deposit feeder in seagrass habitats. Indeed, it is thought that below-ground seagrass production may also foster infaunal secondary production (Orth et al. 1984, Williams and Heck 2001).

Despite limitations, this study highlights some potential links involving landscape configuration, habitat structure, and macrofaunal associations which can be further addressed using hypothesis-driven research. Of course, the generality of this study is limited by the lack of interspersion of seagrass landscape types. Furthermore, potential complexity of relationships involving multiple spatiotemporal scales ob-

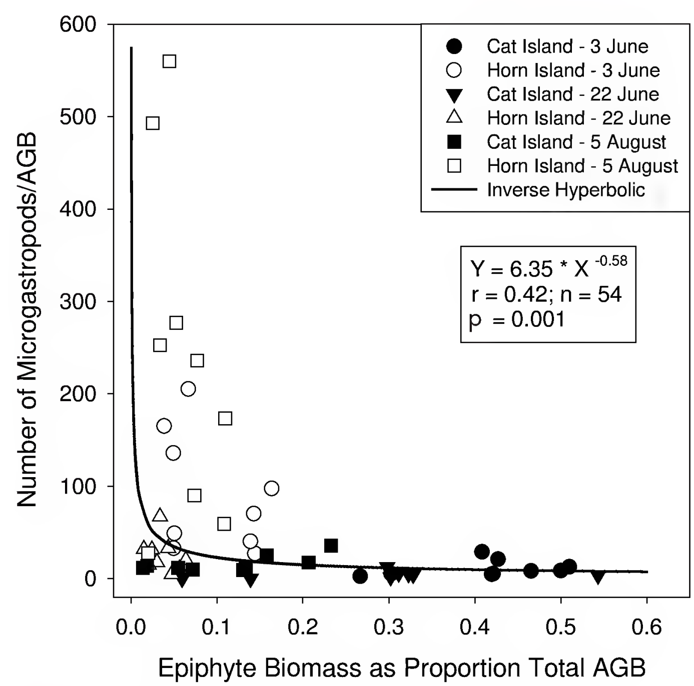

Figure 5.
Inverse hyperbolic relationship between the abundance of microgastropod snails normalized to AGB versus epiphyte biomass expressed as the proportion of total AGB.

scures progress toward a coherent seagrass landscape-habitat function paradigm. Such a paradigm is especially important for successful seagrass restoration, which is predicated on knowing the right abiotic and biotic conditions for the re-establishment of the entire plant and faunal community (Fonseca et al. 1998, Pranovi et al. 2000). Such formidable challenges can only be met with experimental studies of specific mechanisms and effects that are relevant on the landscape scale.

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Literature Cited

Allen, T.F.H. and T.B. Starr. 1988. Hierarchy: Perspectives for ecological complexity. University of Chicago Press, Chicago. 310 p.

Bell, S.S., R.A. Brooks, B.D. Robbins, M.S. Fonseca, and M.O. Hall. 2001. Faunal response to fragmentation in seagrass habitats: implications for seagrass conservation. Biological Conservation 100:115-123.

Bologna, P.A.X. and K.L. Heck. 1999. Macrofaunal associations with seagrass epiphytes: Relative importance of trophic and structural characteristics. Journal of Experimental Marine Biology and Ecology 242:21-39.

Boström, C., E.L. Jackson, and C.A. Simenstad. 2006. Seagrass landscapes and their effects on associated fauna: A review. Estuarine, Coastal and Shelf Science 68:383-403.

Connolly, R.M. and J.S. Hindell. 2006. Review of nekton patterns and ecological processes in seagrass landscapes. Estuarine, Coastal and Shelf Science 68:433-444.

Duffy, J.E., K.S. MacDonald, J.M. Rhode, and J.D. Parker. 2001. Grazer diversity, functional redundancy, and productivity in seagrass beds: an experimental test. Ecology 82:2417-2434.

Edgar, G.J. 1990. Predator-prey interactions in seagrass beds: III. Impacts of the western rock lobster *Panulirus cygnus* George on epifaunal gastropod populations. Journal of Experimental Marine Biology and Ecology 139:33-42.

Fahrig, L. 2002. Effect of habitat fragmentation on the extinction threshold: a synthesis. Ecological Applications 12:346-353.

Fong, C.W., S.Y. Lee, and R.S.S. Wu. 2000. The effects of epiphytic algae and their grazers on the intertidal seagrass *Zostera japonica*. Aquatic Botany 67:251 261.

Fonseca, M.S. and S.S. Bell. 1998. Influence of physical setting on seagrass landscapes near Beaufort, North Carolina, USA. Marine Ecology Progress Series 171:109-121.

Fonseca, M.S., W.J. Kenworthy, and G.W. Thayer. 1998. Guidelines for the conservation and restoration of seagrasses in the United States and adjacent waters. National Oceanic and Atmospheric Administration Coastal Ocean Program Decision Analysis Series, No. 12. National Oceanic and Atmospheric Administration Coastal Ocean Office, Silver Spring, MD. 222 p.

Frost, M.T., A.A. Rowden, and M.L. Attrill. 1999. Effect of habitat fragmentation on the macroinvertebrate infaunal communities associated with the seagrass *Zostera marina L.* Aquatic Conservation: Marine and Freshwater Ecosystems 9:255-263.

Hovel, K.A. and R.N. Lipcius. 2001. Habitat fragmentation in a seagrass landscape: patch size and complexity control blue crab survival. Ecology 82:1814-1829.

Hovel, K.A., M.S. Fonseca, D. Myer, W. Kenworthy, and P. Whitfield. 2002. Effects of seagrass landscape structure, structural complexity and hydrodynamic regime on macrofaunal densities in North Carolina seagrass beds. Marine Ecology Progress Series 243:11-24.

Hughes, A.R., K.J. Bando, L.F. Rodriguez, and S.L. Williams. 2004. Relative effects of grazers and nutrients on seagrasses: a meta-analysis approach. Marine Ecology Progress Series 282:87–99.

Jernakoff, P., A. Brearley, and J. Nielson. 1996. Factors affecting grazer-epiphyte interactions in temperate seagrass meadows. Oceanography and Marine Biology: an Annual Review 34:109-162.

Kovach, W.L. 1994. Oriana for Windows, Version 1.0 User's Manual. Kovach Computing Services, Pentraeth, Wales, UK, 73 p.

Monkkonen, M. and P. Reunanen. 1999. On critical thresholds in landscape connectivity: a management perspective. Oikos 84:302-305.

Moore, T.N. and P.G. Fairweather. 2006. Lack of significant change in epiphyte biomass with increasing extent of measurement within seagrass meadows. Estuarine, Coastal and Shelf Science 68:413-420.

Murphey, P.L. and M.S. Fonseca. 1995. Role of high and low energy seagrass beds as nursery areas for *Penaeus duorarum*

in North Carolina. Marine Ecology Progress Series 121:91-98.

Neckles, H.A., E.T. Koepfler, L.W. Haas, R.L. Wetzel, and R.J. Orth. 1993. Relative effects of nutrient enrichment and grazing on epiphyte-macrophyte (*Zostera marina. L.*) dynamics. Oecologia 93:285-295.

Nelson, T.A. 1997. Interannual variance in a subtidal eelgrass community. Aquatic Botany 56:245-252.

Nelson, W.G. and M.A. Capone. 1990. Experimental studies of predation on polychaetes associated with seagrass beds. Estuaries 13:51-58.

Orth, R.J., K.L. Heck Jr., and J. van Montfrans. 1984. Faunal communities in seagrass beds: A review of the influence of plant structure and prey characteristics on predator-prey relationships. Estuaries 7A:339-350.

Peterson, B.J. and K.L. Heck Jr. 2001. Positive interactions between suspension-feeding bivalves and seagrass - a facultative mutualism. Marine Ecology Progress Series 213:143-155.

Pinckney, J.L. and F. Micheli. 1998. Microalgae on seagrass mimics: Does epiphyte community structure differ from live seagrasses? Journal of Experimental Marine Biology and Ecology 221:59-70.

Pittman, S.J., C.A. McAlpine, and K.M. Pittman. 2004. Linking fish and prawns to their environment: a hierarchical landscape approach. Marine Ecology Progress Series 283:233-254.

Pranovi F., D. Curiel, A. Rismondo, M. Marzocchi, and M. Scattolin. 2000. Variations of the macrobenthic community in a seagrass transplanted area of the Lagoon of Venice. Scientia Marina 64:303-310.

Qurban, M. 2000. The population dynamics and life history of *Bittiolum varium* (Gastropoda: Cerithiidae). M.S., Thesis, Florida Institute of Technology, Melbourne, FL, 86 p.

Sand-Jensen, K. 1977. Effect of epiphytes on eelgrass photosynthesis. Aquatic Botany 3:55–63.

Sand-Jensen, K., N.P. Revsbech, and B.B. Jorgensen. 1985. Microprofiles of oxygen in epiphyte communities on submerged macrophytes. Marine Biology 89:55–62

Sommer, U. 1999. Periphyton architecture and susceptibility to grazing by periwinkles (*Littorina littorea*, Gastropoda). International Revue Hydrobiologie 84:197-204.

SPSS. 2004. SPSS for Windows. Release 13.0. SPSS, Chicago, Illinois, USA.

Turner, S.J., J.E. Hewitt, and G. Funnell. 1999. Seagrass patches and landscapes: The influence of wind-wave dynamics and hierarchical arrangements of spatial structure on macrofaunal seagrass communities. Estuaries 22:1016-1032.

van Montfrans, J., R.J. Orth, and S.A. Vay. 1982. Preliminary studies of grazing by *Bittium varium* on eelgrass periphyton. Aquatic Botany 14:75-89.

Verbeke, G. and G. Molenberghs. 2000. Linear Mixed Models for Longitudinal Data. Springer-Verlag. New York, 568 p.

Williams, S.L. and K.L. Heck Jr. 2001. Seagrass community ecology. In: M.D. Bertness, S.D. Gaines, and M.E. Hay eds. Marine Community Ecology, Sinauer Associates, Inc., Sunderland NJ, USA, p.317-337.

With, K.A. and T.O. Crist. 1995. Critical thresholds in species' responses to landscape structure. Ecology 76:2446-2459.

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SEAGRASS DISTRIBUTION IN THE PENSACOLA BAY SYSTEM, NORTHWEST FLORIDA

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ABSTRACT: Aerial surveys of seagrass coverage in the Pensacola Bay system (PBS) have been conducted during 1960, 1980, 1992 and 2003. This report summarizes the results for the 2003 survey and compares the results to those previously reported for other surveys. The estimated coverage of seagrass for the PBS during 2003 was 1,654 ha. Continuous and patchy coverages ranged from 0 to 684 ha and 11 to 543 ha, respectively, for five PBS subsystems. In 2003, the majority of seagrass coverage occurred in Santa Rosa Sound (76%). Declines in total coverage occurred for East Bay (93%) and Escambia Bay (75%) whereas increases were observed for Pensacola Bay (32%) and Santa Rosa Sound (8%). The approximate 9% decline (about 160 ha or 395 a) in total coverage since 1992 represents an estimated 7 to 8 million dollar loss in ecological services. The changes in coverage are likely due to naturally occurring and anthropogenic factors but it is not possible to differentiate the relative contributions of these factors alone and in combination on seagrass distribution. The ability of seagrasses to exist long-term in Florida's fourth largest estuarine system is uncertain due to the adverse effects of rapid urbanization in the watershed. Active resource management which includes more frequent *in-situ* monitoring and aerial assessment and the availability of relevant water and sediment quality criteria protective of submerged aquatic vegetation are needed to prevent future declines.

Introduction

Seagrasses have at least 13 ecological roles (Dawes et al. 2004) and they support a diverse biotic community that may contain as many as 113 species of epiphytes, 148 macroalgal species, 80 macrofaunal species, and 75 fish species (Humm 1964, Virnstein et al.1983, Zieman and Zieman 1989, DeTroch et al. 1996). Seagrass meadows including those dominated by *Thalassia testudinum* Konig (turtle grass), the most abundant species in the Pensacola Bay System (PBS), supported twice the macrofauna than did unvegetated sediments (Santos and Simon 1974, Virnstein et al. 1983). The economic importance of one seagrass acre has been estimated to be between $9,000 and $28,000 (Texas) and $20,500 (Florida) due to commercial, recreational and storm protection functions (Handley et al. 2007) and $19,000 based on nutrient cycling (Costanza et al. 1997). About 1.2 million of 59 million hectares (ha) of seagrasses have been destroyed worldwide during the last decade and, similar to corals, seagrasses are considered to be in a crisis stage. Declines have occurred at 40 locations (Short and Wylie-Echeverria 1996, Hemminga and Duarte 2000) including the Gulf of Mexico (GOM), where seagrass coverage has been reduced 20 to 100% during the past 50 yrs (Dawes et al. 2004, USGS and GOMP 2004). For example, about 85% of the seagrasses along Florida's coasts have been destroyed by 1992 (USEPA 1992).

Florida's Gulf Coast contains about 680,000 ha of seagrasses of which about 2% (17,474 ha) is adjacent to the Florida panhandle region (Madley et al. 2003). The trends in coverage and condition of the seagrasses in northwest Florida are considered to be poorly understood (Dawes et al. 2004). However, seagrass research for the PBS, the focus of this report, has been conducted intermittently during the past 40 yrs (Table 1) and includes four major aerial surveys conducted since the 1960s. The objectives of this report are to summarize the results of the 2003 aerial survey and to compare the results primarily to those for the previous survey conducted during 1992.

Materials and Methods

Study Area

The PBS is located in northwest Florida and its watershed consists of about 18,000 km^2 of forests, agricultural lands, and urban and industrial areas (Figure 1). It is the fourth largest estuary in Florida and extends 32 km inland and comprises about 886 km of coastline and 435 km of inland waterways. Mean water residence time is about 25 d (Solis and Powell 1999). The PBS is comprised of five subsystems: Big Lagoon, East Bay, Escambia Bay, Pensacola Bay, and Santa Rosa Sound (Figure 1, Table 2). Santa Rosa Sound and Big Lagoon are marine lagoons that are parallel to the GOM and retain high salinities due to limited freshwater input. Both areas contain sections that are classified as Outstanding Florida Waters (FDEP 2001).

Aerial Surveys

Although aerial surveys of the PBS are available since the 1940s, only those conducted during 1960, 1980, 1992 and 2003 were specific for determining seagrass coverage. Results of the 1960, 1980, and 1992 surveys have been report-

ed previously (FDEP 2001, Schwenning et al. 2007) and the results for the October 2003 survey form the basis of this report. The experimental techniques for these surveys have differed, and these differences need to be considered in the context of the conclusions for the across-year comparisons reported here. Methodologies for the 1992 and 2003 surveys were similar but not identical. The seagrass data were derived from interpretation of 1:24,000 natural color scale photographs (1992 survey) and 1:12,000 scale color infrared photographs (2003 survey). Personnel from NASA (Stennis, MS) and Aerial Cartographics of America (Orlando, FL) performed the 1992 and 2003 flights, respectively. Black and white photographs (1:24,000 scale) and natural color photographs (1:24,000 scale) were used for the 1960 and 1980 surveys, respectively. Personnel from U.S. Geological Survey's National Wetlands Research Center (NWRC, Lafayette, LA) conducted the mapping procedure which included photo-interpretation of the aerial photographs, cartographic transfer, and digitization for all surveys. The classification scheme for all surveys was derived by USGS/NWRC based on the coastal land cover classification system of the National Oceanic and Atmospheric Administration's Coastwatch Change Analysis Project (NOAA 2003). The amount of groundtruthing or field verification of the aerial surveys has been variable, and for the 2003 survey it consisted of single visits to the five PBS subsystems to confirm seagrass presence. No *in-situ* measurements of plant condition or species identification were performed.

Results and Discussion

Seagrass Coverage

An estimated 1,654 seagrass-vegetated ha (4,085 acres) were present in the PBS based on the October 2003 aerial survey (Table 3). Of this total, most coverage occurred in Santa Rosa Sound (76%). Seagrass coverage as a percent of total surface area was 18% (Santa Rosa Sound), 5% (Big Lagoon), 1% (Pensacola Bay) and < 1% (Escambia Bay, East Bay). Continuous and patchy coverages ranged from 0 to 684 ha and 11 to 543 ha, respectively in the PBS (Table 3). About 52% of the 2003 total coverage was continuous, and almost all continuous coverage (99%) occurred in Santa Rosa Sound and Big Lagoon.

The 2003 total seagrass coverage (1,654 ha) represents an almost 9% reduction relative to the estimated total for the 1992 survey (1,814 ha). Reductions in total coverage relative to 1992 occurred for East Bay (93%), and Escambia Bay (75%) while increases were observed for Pensacola Bay (32%) and Santa Rosa Sound (8%) (Figure 2). Total coverages in Big Lagoon for 1992 and 2003 were almost identical. Continuous coverage decreased in Escambia Bay (69%), Santa Rosa Sound (14%), Pensacola Bay (100%), and East Bay (100%), which contrasted an almost 64% increase in coverage for Big Lagoon. Patchy coverage decreased in Big

TABLE 1. *Listing of seagrass and habitat condition research conducted for the Pensacola Bay System, Florida.*

Subsystem	References
Big Lagoon	Hopkins 1973 Heck et al. 1996 Lores et al. 2000 FDEP 2001
East Bay	Van Breedveld 1966 McNulty et al. 1972 Rogers and Blisterfield 1975
Escambia Bay	Moore 1963 USDOI 1970 Livingston et al. 1971 McNulty et al. 1972 Hopkins 1973 Rogers 1974 Rogers and Blisterfield 1975 Woodward and Clyde 1997 Lewis et al. 2000 Lores et al. 2000 Lores and Sprecht 2001 Murrell et al. 2002 Murrell and Lores 2004
Pensacola Bay	Moore 1963 McNulty et al. 1972 Rogers and Blisterfield 1975 Rodriguez and Hunner 1994 Murrell et al. 2002
Santa Rosa Sound	Moore 1963 Van Breedveld 1966 McNulty et al. 1972 Hopkins 1973 Rogers and Blisterfield 1975 Winter 1978 Macauley et al. 1988 Heck et al. 1996 Lores et al. 2000 FDEP 2001 Lewis et al. 2001, 2002
Pensacola Bay System (general)	USEPA 1975 Williams 1981 Lewis 1986 Ridenauer and Shambaugh 1986 Jones et al. 1992 Collard 1991 Thorpe et al. 1997 DeBusk et al. 2002 Dawes et al. 2004 McRae et al. 2004 USEPA 2004 USEPA 2005 Schwenning et al. 2007

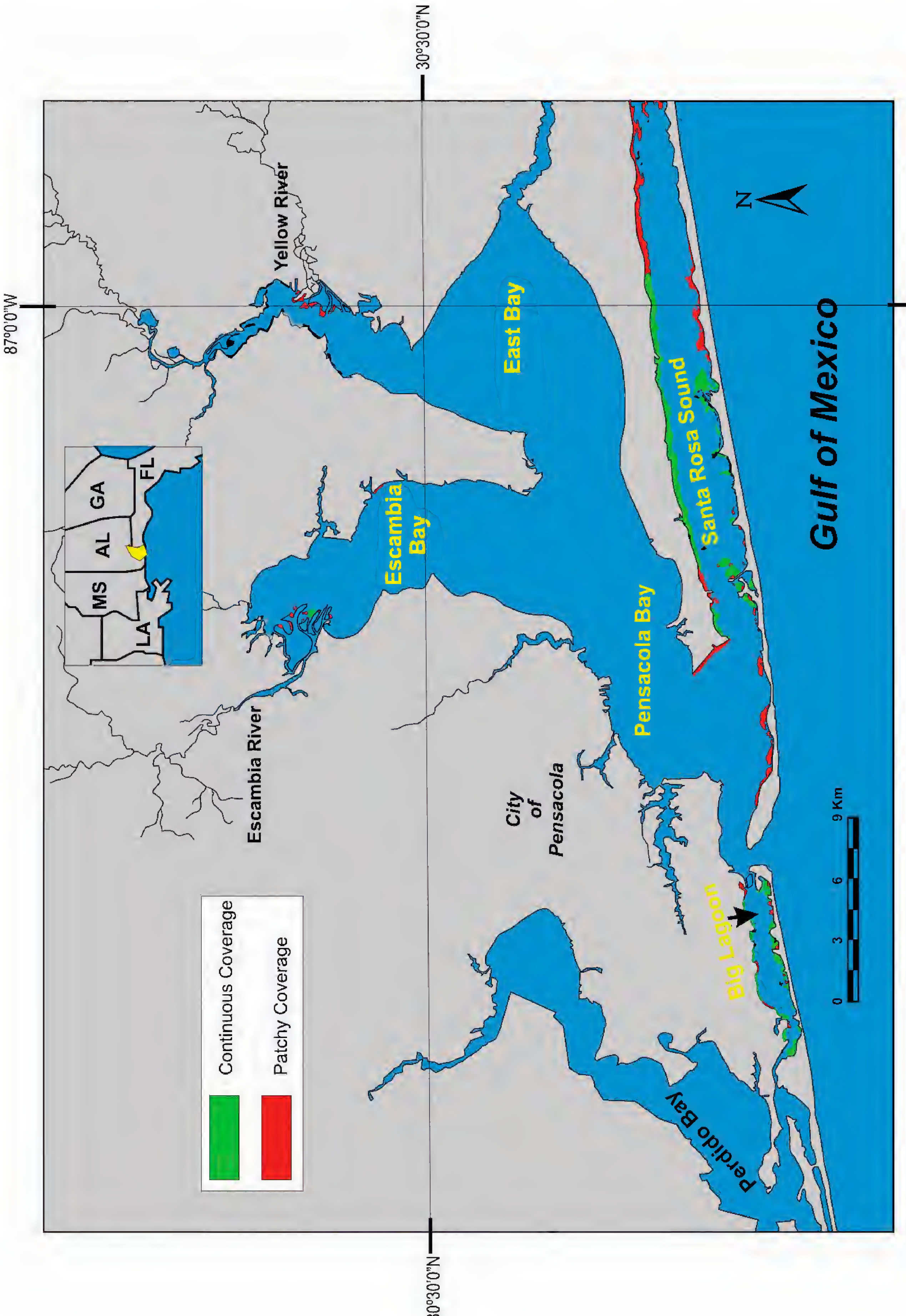

Figure 1.
The Pensacola Bay System in northwest Florida and its watershed (insert). Shaded areas represent continuous (green) and patchy (red) seagrass coverage estimated from a 2003 aerial survey.

TABLE 2. *Geomorphological characteristics and water body classifications for Pensacola Bay subsystems. Some data adapted from USEPA (1975). N/A – not available.*

Subsystem	**Surface Area (km^2)**	**Volume (10^6 m^3)**	**Mean Depth (m)**	**Water Body Classification[1]**	**Florida Impaired Water[2]**	**Assessed Parameters**
Big Lagoon	46.8	N/A	N/A	2 (OFW)	N	-
East Bay	25.9	259.3	2.4	2	Y	Fecal coliforms, Nutrients
Escambia Bay	57	225.7	2.4	3M	Y	Nutrients
Pensacola Bay	133.6	793.8	5.9	3M	Y	Bacteria
Santa Rosa Sound	68.2	N/A	N/A	2 (OFW)	Y	Fecal coliforms

[1] 2 = shellfish propagation or harvesting, 3M = Recreation, propagation and maintenance of well-balanced populations of fish and wildlife (FDEP 1996). OFW = Portion designated as Outstanding Florida Water (OFW) (FDEP 2004).

[2] Contains segments for TMDL development (FDEP 2006).

Lagoon (51%), East Bay (93%), and Escambia Bay (76%) and increased in Pensacola Bay (50%) and Santa Rosa Sound (58%).

Historical Perspective

The numerous seagrass investigations conducted in the PBS have been temporally and spatially inconsistent and research methodologies have varied. Seagrass mapping has been the focus of most studies, and only limited information is available describing physiological and morphological parameters of plant condition such as above- and below-ground biomass, shoot density, blade height and epiphyte/biomass ratio, which have been reported for grasses in Big Lagoon, Santa Rosa Sound and Escambia River delta region (Heck et al. 1996, Lores et al. 2000, FDEP 2001).

Historically, the largest seagrass declines within the PBS occurred between 1960 and 1980 and subsequent declines have been less severe (Table 3, Figure 2). Seagrass meadows in Santa Rosa Sound and Big Lagoon, with few exceptions, have dominated the PBS since 1960 (coverage range as a percent of total = 75 to 90%). Their combined dominance increased 12% since 1992 to 87% of total coverage in 2003. The consistent coverages in these areas are due, at least in part, to relatively less urbanization of the shorelines (sections are included in a state park and a national seashore) and to limited freshwater input which stabilizes salinity and reduces the entry of watershed contaminants. In a detailed study conducted in these areas (Heck et al. 1996), seagrasses declined during 1993 – 1995 in both areas due to a combination of propeller scarring, reduced water clarity, changes in salinity and burial due to hurricanes. Lores et al. (2000) reported seagrass coverage was also decreasing in Big Lagoon during 1997 and 1998.

Seagrass research has been conducted less frequently in East, Escambia and Pensacola Bays, where seagrasses have been relatively uncommon since the 1960s (range of combined total coverages = 9 to 25%; Table 3). Seagrass coverage in East Bay was almost non-detectable and the least of any subsystem in 2003. Submerged aquatic vegetation in Escambia Bay is primarily limited to tidal freshwater grasses, *Vallisneria americana* Michx (tape grass) in the upper reaches and the more salinity tolerant *Ruppia maritima* L. (wigeon grass) in more seaward areas. *Vallisneria americana* is not a true seagrass, although its ecological value is similar to that of seagrasses. This species is included in this analysis, since prior assessments did not differentiate between species. Total coverage in Escambia Bay fluctuates greatly, with a

TABLE 3. *Continuous (C) and patchy (P) seagrass coverage (hectares) in Pensacola Bay subsystems. Data for 1960, 1980 and 1992 aerial surveys from Schwenning et al. (2007).*

		AERIAL SURVEY			
Subsystem		**1960**	**1980**	**1992**	**2003**
Big Lagoon	C	107	193	99	162
	P	164	43	118	58
East Bay	C	45	12	5	0
	P	431	87	160	11
Escambia Bay	C	4	4	36	11
	P	101	20	143	34
Pensacola Bay	C	44	10	13	0
	P	328	46	101	151
Santa Rosa Sound	C	1,247	850	796	684
	P	1,387	629	343	543
TOTAL		**3,858**	**1,894**	**1,814**	**1,654**

historical maximum occuring in 1992 followed by a 75% decline in 2003. Lores et al. (2000) reported increased coverage of *V. americana* near the Escambia River delta for 1997 and 1998 relative to 1992, but coverage decreased during 2000 due to drought and high salinity (Lores and Sprecht 2001). Although continuous seagrass coverage in Pensacola Bay was not detectable in 2003, patchy coverage increased from 101 ha to 151 ha since 1992.

Causes of Seagrass Declines

The factors responsible for declines in seagrass coverage for the PBS, other than direct effects due to mechanical and physical factors, have been more often speculative than confirmed. Supportive documentation is limited, and current understanding of the causative factors responsible for the seagrass changes alone and in combination remains elusive, which has limited effective restoration efforts and resource management. Declines have been attributed to controllable factors such as point and nonpoint source contaminants, prop scarring, dock shading, armoured shorelines, and dredging, as well as to the non-controllable effects of episodic weather events which have been increasing in recent years. A few investigators have reported the effects of nutrients (Heck et al. 1996, Lores et al. 2001) and low salinity (Lores and Specht 2001) and the potential for chemical phytotoxicity (Lewis et al. 2007) on seagrasses within the PBS.

The PBS is a contaminant-impacted estuary based on the results of many reports included in Table 1. Turbidity due to erosion and accelerated eutrophication have reduced light penetration in all subsystems. In addition, regulatory effects-based guidelines and criteria to protect marine life in water and sediment have been commonly exceeded in several subsystems for several non-nutrient contaminants (for example, DeBusk et al. 2002, USEPA 2005), suggesting a potential for toxicity. This is the case for Escambia Bay, which is the most contaminated subsystem and a priority site for conservation (Beck et al. 2000). In contrast, Santa Rosa Sound is the least chemically contaminated area within the PBS (FDEP 2001, Lewis et al. 2007).

Despite differences in environmental conditions among subsystems, it remains to be determined if the sensitivities of seagrasses and other submerged aquatic vegetation to non-nutrient contaminants is equivalent to those for zoobenthos and fish for which most regulatory numerical guidelines and criteria have been developed. In addition to this uncertainty is the continued absence of national and state numerical nutrient criteria which further hinders effective management of these marine angiosperms.

Summary and Recommendations

Despite the limitations and sources of error associated with aerial seagrass photography (see Carlson and Madley 2007), it was clear that there has been no net gain in seagrass coverage within the PBS since 1992. The few site-specific gains in coverage were overshadowed by declines in other subsystems which resulted in an estimated net loss of about 9% or 160 ha (~ 395 acres) system-wide. The relevance of the 2003 coverage to current coverage (2007) is unknown. Two major hurricanes, Ivan (2004) and Dennis (2005) made almost direct landfall near the PBS. Their impacts have not been reported in the scientific literature, nor have impacts associated with the ongoing urbanization of the PBS shoreline that has occurred since 2003.

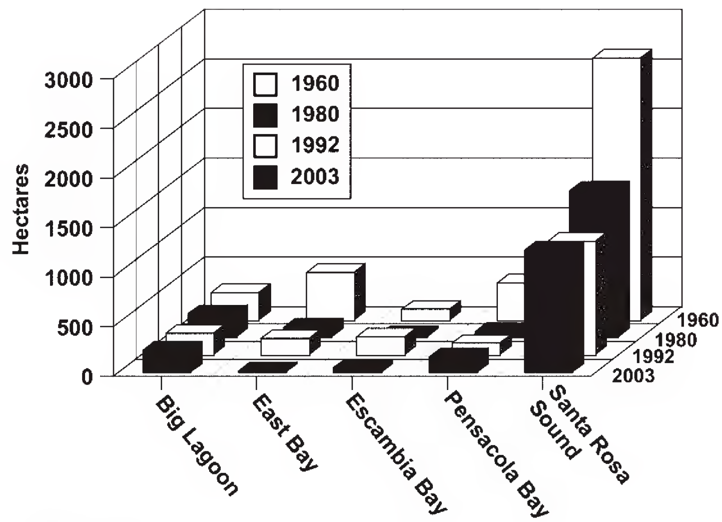

Figure 2.
Total seagrass coverage (hectares) for five subsystems in the Pensacola Bay System based on four aerial surveys. Data for 1960, 1980 and 1992 surveys from Schwenning et al. (2007).

The rapid urbanization of the PBS watershed is expected to continue. The populations in Escambia and Santa Rosa Counties is predicted to increase 20% and 64%, respectively, by 2020 (Zwick and Carr 2006). The magnitude of this urban transformation on near-shore seagrasses is unknown. This uncertainty will remain until an effective and long-term resource management plan is implemented, particularly for near-shore areas containing extensive seagrass coverage such as Santa Rosa Sound. Several management plans have been proposed (Rogers and Blisterfield 1975, Collard 1991, FDEP 2001), and these should include frequent aerial assessments (every 3 to 5 yrs) to determine coverage and more frequent *in-situ* evaluation to determine plant condition. These assessments are important so that in the long-term biocriteria predictive of habitat quality can be developed. Other management considerations should include promotion of shoreline configurations supportive of seagrass meadows and increased efforts to control coastal entry of non-point source contaminants. Of additional importance is the need to establish a separate designated use category to protect coastal submerged vegetation as well as the development of supportive regulatory numerical criteria for common near-shore contaminants and those of emerging concern (Daughton 2005).

Finally, assessment of the economical value of ecosys-

tem services has become increasingly important to the environmental policy and management process (Costanza et al. 1997, Carpenter and Turner 2000, USEPA 2006). The loss of ecological services associated with the estimated 395 acre decline in seagrass coverage for the PBS since 1992 represents an approximate $8.1 million impact ($20,500/acre; Handley et al. 2007) or, if based on nutrient cycling, a $7.5 million loss ($19,000/acre; Costanza et al. 1997).

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Literature Cited

Beck, M.W., M. Odaya, J.J. Bachant, J. Bergan, B. Keller, R. Martin, R. Mathews, C. Porter, and G. Ramseur. 2000. Identification of priority sites for conservation in the northern Gulf of Mexico: an ecoregional plan. The Nature Conservancy, Arlington, VA, USA. 47 p.

Carlson, P.R. and K. Madley. 2007. State of Florida. In: L. Handley, D. Alteman, and R. DeMay, eds. Seagrass Status and Trends in the Northern Gulf of Mexico: 1940 – 2002. U.S. Geological Survey Scientific Investigations Report 2006 – 5287 and U.S. Environmental Protection Agency 855-R-04-003, U.S. Geological Survey, Reston, VA, USA, p 98-114.

Carpenter, S.R. and M. Turner. 2000. Opening the black boxes: ecosystem science and economic evaluation. Ecosystems 3: 1-3.

Collard, S. 1991. Management options for the Pensacola Bay System: the potential value of seagrass transplanting and oyster bed refurbishment programs. Water Resources Special Report 91-4. Northwest Florida Water Management District, Havana, FL, USA, 33 p.

Costanza, R., R. d'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limberg, S. Naeem, R.V. O'Neill, J. Parnelo, R.G. Raskin, P. Sutton, and M. van den Belt. 1997. The value of the world's ecosystem services and natural capital. Nature 387:253-260.

Dawes, C.J., R.C. Phillips, and G. Morrison. 2004. Seagrass communities of the Gulf Coast of Florida: Status and Ecology. Tampa Bay Estuary Program, St. Petersburg, FL, USA, 74 p.

Daughton, C.G. 2005. Emerging chemicals as pollutants in the environment: a 21st century perspective. Renewable Resources Journal 23:6-23.

DeBusk, W.F., I. Poyer, and I. Herzfeld. 2002. Sediment quality in the Pensacola Bay System. Final Report. Northwest Florida Water Management District, Havana, FL, USA, 66 p.

DeTroch, M., J. Mees, I. Papadopoulos, and E.O. Wakwabi. 1996. Fish communities in a tropical bay (Gazi Bay, Kenya): seagrass beds vs. unvegetated areas. Netherlands Journal of Zoology 46:236-252.

Florida Department of Environmental Protection (FDEP). 1996. Chapter 62-302 of the Florida Administration Code Rule: 62-302, 530 (45), Tallahassee, FL, USA.

Florida Department of Environmental Protection (FDEP). 2001. Seagrass management plan for Big Lagoon and Santa Rosa Sound. Florida Department of Environmental Protection, Pensacola, FL, USA, 44 p.

Florida Department of Environmental Protection (FDEP). 2004. Integrated water quality assessment for Florida: 2004, 305 (b) report and 303 (d) list update. Florida Department of Environmental Protection, Tallahassee, FL, USA, 190 p.

Florida Department of Environmental Protection (FDEP). 2006. Pensacola (Group 4) verified list of impaired waters. Available on-line at URL: http://dep.state.fl.us/water/tmdl.

Handley, L., D. Altsman, and R. DeMay, eds. 2007. Seagrass Status and Trends in the Northern Gulf of Mexico: 1940-2002. U.S. Geological Survey Scientific Investigations Report and U.S. Environmental Protection Agency 855-R-04-003, U.S. Geological Survey, Reston, VA, USA, 267 p.

Heck, K.L. Jr., M.J. Sullivan, J.M. Zande and C.A. Moncreiff. 1996. An ecological analysis of seagrass meadows of the Gulf Island National Seashore - years one, two, and three: seasonal assessment and inventory, interaction studies and assessment/inventory. Marine Environment Science Consortium, University of South Alabama, Dauphin Island, AL, USA, 88 p.

Hemminga, M.A. and C.M. Duarte. 2000. Seagrass Ecology. Cambridge University Press, Cambridge, MA, USA, 326 p.

Hopkins, T.S. 1973. Marine Ecology in Escarosa. Coastal Coordinating Council. Contract No. CCC-03-72 and CCC-14-72. Florida Department of Natural Resources, Tallahassee, FL, USA, 100 p.

Humm, H.J. 1964. Epiphytes of the seagrass *Thalassia testudinum* in Florida. Bulletin of Marine Science of the Gulf and Caribbean 14:306-341.

Jones, W.K., J.H. Cason, and R. Bjorklund. 1992. A literature-based review of the physical, sedimentary and water quality aspects of the Pensacola Bay System. Water Resources Special Report 92 - 5, Northwest Florida Water Management District, Havana, FL, USA, 58 p.

Lewis, T. 1986. The Pensacola Bay System. An analysis of estuarine and degradations and their relationships to land management practices. Florida Department of Community Affairs, Tallahassee, FL, USA, 132 p.

Lewis, M.A., D.E. Weber, and B. Albrecht. 2000. Quality and toxicity in Gulf of Mexico coastal areas below ten wastewater discharges. Environmental Toxicology and Chemistry 19:192-203.

Lewis, M.A., S.S. Foss, P.S. Harris, R.S. Stanley, and J.C. Moore. 2001. Sediment chemical contamination and toxicity associated with a coastal golf course complex. Environmental Toxicology and Chemistry 20:1390-1398.

Lewis, M.A., R. Boustany, D. Dantin, and R. Quarles. 2002. Effect of a coastal golf course complex on water quality and nutrient bioavailability to the Gulf of Mexico. Ecotoxicology and Environmental Safety 53:154-162.

Lewis, M.A., D. Dantin, K. Abel, and C. Lewis. 2007. Florida seagrass habitat evaluation: A comparative survey for chemical quality. Environmental Pollution 146:206-218.

Livingston, R.J. 1971. The effects of dredging and eutropication on Mulat-Mulatto Bayou (Escambia Bay, Pensacola, Florida). Report for Florida Department of Transportation. F.S.U. Research No. 111308016, Tallahassee, FL, 40 p.

Lores, E.M., E. Pasko, J. Patrick, R.L. Quarles, J. Campbell, and J. Macauley. 2000. Mapping and monitoring of submerged aquatic vegetation in Escambia - Pensacola Bay System, Florida. Gulf of Mexico Science 18:1-14.

Lores, E.M. and D.T. Specht. 2001. Drought-induced decline of submerged aquatic vegetation in Escambia Bay, Florida. Gulf of Mexico Science 19:161-164.

Macauley, J.M., J.R. Clark, and W.A. Price. 1988. Seasonal changes in the standing crop and chlorophyll content of *Thalassia testudinum* Banks ex König and its epiphytes in the northern Gulf of Mexico. Aquatic Botany 31:277-287.

Madley, K.A., J.M. Lewis, J.D. Krolick, and E.C. Santi. 2003. Florida Marine Research Institute website seagrass pages. Available online at URL: http://floridamarine.org.

McNulty, J.K., W.N. Lindall, and J.E. Sykes. 1972. Cooperative Gulf of Mexico estuaries inventory and study, Florida: phase I, area description. National Oceanic and Atmospheric Administration Technical Report Circ-368, U.S. Department of Commerce, Seattle, WA, USA, 126 p.

McRae, G., M. Julian, and H. Brown. 2004. Florida's inshore marine monitoring and assessment program (IMAP). Annual Report, Year Five. FWRI File Code: F2248-99-A4. Fish and Wildlife Research Institute, St. Petersburg, FL, USA, 93 p.

Moore, D.R. 1963. Distribution of the seagrass, *Thalassia,* in the United States. Bulletin of Marine Science of the Gulf and Caribbean 13:329-342.

Murrell, M.C., R.S. Stanley, E.M. Lores, G.T. DiDonato, L.M. Smith, and D.A. Flemer. 2002. Evidence that phosphorus limits phytoplankton growth in a Gulf of Mexico estuary: Pensacola Bay, FL, USA. Bulletin of Marine Science 70:155-167.

Murrell, M.C. and E.M. Lores. 2004. Phytoplankton and zooplankton seasonal dynamics in a subtropical estuary: importance of cyanobacteria. Journal of Plankton Research 26:371-382.

National Oceanic and Atmospheric Administration (NOAA). 2003. NOAA coastal change analysis program (C-CAP): guidance for regional implementation. Available on-line at URL: http://csc.noaa.gov/crs/.

Ridenauer, J. and C. Shambaugh. 1986. An analysis of estuarine degradations within the Pensacola Bay System and their relationship to land management practices. Florida Department of Community Affairs, Tallahassee, FL, USA, 132 p.

Rodrigeuz, G. and W.C. Hunner. 1994. Water quality sampling data report of Pensacola Bay for March 1993 - February 1994. Technical File Report 94-4, Northwest Florida Water Management District, Havana, FL, USA, 392 p.

Rogers, R.G. 1974. Seagrass revegetation in Escambia Bay, Florida. In: Proceedings of the First Annual Conference on Restoration of Coastal Vegetation in Florida, Tampa, p 21-25.

Rogers, R.G. and F.T. Blisterfield. 1975. Loss of submerged vegetation in the Pensacola Bay system 1949 - 1974. In: R.R. Lewis, ed., Proceedings of the Second Annual Conference on Restoration of Coastal Vegetation in Florida. Hillsborough Community College, Tampa, FL, USA, p 36-50.

Santos, S.L. and J.L. Simon. 1974. Distribution and abundance of the polychaetous annelids in a south Florida estuary. Bulletin of Marine Science 24:669-689.

Schwenning, L., T. Bruce, and L.R. Handley. 2007. Pensacola Bay System. In: L. Handley, D. Altsman, and R. DeMay eds., Gulf of Mexico Seagrass Status and Trends. U.S. Geological Survey Scientific Investigations Report, 2006 – 5287 and U.S. Environmental Protection Agency 855-R-04-003, U.S. Geological Survey, Reston, VA, USA, p 129-142.

Short, P.T. and S. Wylie-Echeverria. 1996. Natural and human-induced disturbance of seagrasses. Environmental Conservation. 23:17-27.

Solis, R.S. and G.L. Powell. 1999. Hydrography, mixing characteristics and residence times of Gulf of Mexico estuaries. In: T.S. Bianchi, J.R. Pennock, and R.R. Twilley, eds., Biogeochemistry of Gulf of Mexico Estuaries. John Wiley and Sons. Inc., New York, NY, USA, p. 29-62.

Thorpe, P., R. Bartel, P. Ryan, K. Albertson, T. Pratt, and D. Cairns. 1997. The Pensacola Bay system surface water improvement and management plan, Program Development Series 97-2. Northwest Florida Water Management District, Havana, FL, USA, 146 p.

U.S. Department of the Interior (USDOI). 1970. Effects of pollution on water quality in Escambia River and Bay, Florida. U.S. Department of Interior. Federal Water

Pollution Control Administration, Athens, GA, USA. 63 p.

U.S. Environmental Protection Agency (USEPA). 1975. Environmental and Recovery Studies of Escambia Bay and the Pensacola Bay System, Florida. EPA-904/9-76-016, Region IV, Atlanta, GA, USA, 322 p.

U.S. Environmental Protection Agency (USEPA). 1992. Status and trends of emergent and submerged vegetated habitats, Gulf of Mexico, USA. EPA-800-R92-003, Office of Water, Washington, DC, USA, 161 p.

U.S. Environmental Protection Agency (USEPA). 2004. National Coastal Condition Report II. EPA-620/R-031002, Office of Water, Office of Research and Development, Washington, DC, USA, 286 p.

U.S. Environmental Protection Agency (USEPA). 2005. The ecological condition of the Pensacola Bay System. EPA-620/R-05/002, Gulf Ecology Division, Gulf Breeze, FL, USA, 38 p.

U.S. Environmental Protection Agency (USEPA). 2006. Ecological benefits assessment strategic plan. EPA-240-R-06-001, Office of the Administrator, Washington, DC, USA, 32 p.

U.S. Geological Survey (USGS) and Gulf of Mexico Program (GOMP). 2004. Seagrass habitat in the northern Gulf of Mexico: degradation, conservation and restoration of a valuable resource. EPA-855-R-04-001, Gulf of Mexico Program Office, Stennis Space Center, MS, USA, 25 p.

Van Breedveld, J.F. 1966. Internal Report Preliminary Algal Survey. Florida Board of Conservation Marine Laboratory, Maritime Base, St. Petersburg, FL, USA, 48 p.

Virnstein, R.W., P.W. Mikkelson, K.D. Cairns, and M.A. Capone. 1983. Seagrass beds versus sand bottoms: The trophic importance of their associated benthic invertebrates. Florida Scientist 46:363-381.

Williams, D.R. 1981. Marine grass beds inventory, North Florida coast (October 1980). Final report prepared by the U.S. Environmental Protection Agency for the U.S. Fish and Wildlife Service for the USFWS, TS-AMD-80099. U.S.EPA, Cincinnati, OH, USA, 89 p.

Winter, P.A. 1978. Evaluation of Seagrass Resources, Santa Rosa Island, Florida. Final Report to the Army Corps of Engineers. Florida Department of Environmental Regulation, Pensacola, FL, USA, 25 p.

Woodward and Clyde Associates. 1997. Environmental studies in Escambia Bay and Perdido Bay systems. Final Report. Prepared for Champion International Corp., Project No. 97F299, Tallahassee, FL, USA, 210 p.

Zieman, J.C. and R.T. Zieman. 1989. The ecology of the seagrass meadows of the west coast of Florida: a community profile. Biological Report 85 (7.25), Washington, DC, USA, 155 p.

Zwick, P.D. and M.H. Carr. 2006. Florida 2060 – a population distribution scenario for the State of Florida. Final report, Geoplan Center, University of Florida, Gainesville, FL, USA, 25 p.

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VARIABILITY IN ESTIMATING ABUNDANCE OF POSTLARVAL BROWN SHRIMP, *FARFANTEPENAEUS AZTECUS* (IVES), MIGRATING INTO GALVESTON BAY, TEXAS.

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ABSTRACT: Three sets of monitoring data were used to examine the variability associated with abundance estimation of postlarval brown shrimp, *Farfantepenaeus aztecus* (Ives) in Bolivar Roads, Texas—the main connection between the Gulf of Mexico and Galveston Bay. Abundance of postlarvae (PL) caught with Renfro beam trawl varied greatly in different years on the same dates. A "spring peak" of brown shrimp PL migrating into Galveston Bay was found for 2 April with a quadratic regression fit to 6-day moving averages of daily mean abundance from 22 yrs of monitoring data: $Ln(PL+1) = 0.8736 + 0.09037Day - 0.0004934Day^2$ (adj-R^2 = 0.83, n = 159), where Day is Julian day. Abundance varied by four orders of magnitude (0 to 24,616 PL/tow) in just 4 d during a four-week intensive monitoring of PL during the 1987 spring peak. Abundance also varied by three orders of magnitude between the North and South Jetty sites during the same collection time. During a third study, PL abundance varied by two orders of magnitude along 360 m of the beach in < 4 hr. These investigations demonstrate that detecting significant differences in PL shrimp abundance in a pass requires substantial sampling that may not be logistically possible. However, best estimates could be obtained by including as many dates as possible, followed by including more sites, and finally by collecting during both day and night. Conclusions drawn from abundance studies of PL shrimp, fish, and crab immigrants through estuarine passes that are based on only a few samples should be reviewed.

INTRODUCTION

The brown shrimp, *Farfantepenaeus aztecus* (Ives), is a key commercial species in the shrimp fishery of the northwestern Gulf of Mexico (GOM). Most adults inhabit water depths of 20-65 m (Darnell et al. 1983, Neal et al. 1983) and spawning and larval development occur in these waters. Postlarvae (PL) migrate into the bay where they grow for about three months in salt marshes (Zimmerman and Minello 1984). Then, as advanced juveniles or sub-adults, they migrate back through the bays to the GOM, during which time they recruit to the bait and bay shrimp fisheries. All shrimp fisheries are valuable, are managed based on age-0 individuals (J. Nance and F. Patella, pers. comm., NMFS, Galveston, TX), and are characterized by large variability in annual catches (Klima et al. 1986). It is beneficial to commercial shrimp fishers and resource managers to have a forecast of the upcoming harvest, and the abundance of immigrating PL is a potential indicator of shrimp harvest (Baxter 1963, Berry and Baxter 1969, Baxter and Sullivan 1986).

Various attempts to establish an early forecast using PL abundance have been unsuccessful (Williams and Deubler 1968, Berry and Baxter 1969, Sutter and Christmas 1982, DeLancey et al. 1994). These forecasting models have relied upon three important assumptions: (1) mortality rates for young brown shrimp in the estuary are either constant or vary in a regular manner seasonally from year to year; (2) the majority of recruitment of PL shrimp to estuarine nurseries occurs during the same months each year; and (3) accurate estimates of PL immigration to bays and estuaries have been obtained. Mortality rates of juvenile shrimp can be highly variable on a weekly or annual basis, but few measurements of this mortality are available (Minello et al. 1989). Accurate estimates of the influx of PL might not be possible; even the precision of such estimates has been studied only to a limited degree (Berry and Baxter 1969, Caillouet et al. 1968, 1970, Lochmann 1990). Only about 60% of the age-0 shrimp recruit to the fishery during the early summer, the rest recruit mostly during the next four months. The PL for the summer recruitment enter the estuaries in late winter and early spring, and Berry and Baxter (1969) hypothesized that the magnitude of the spring peak immigration might control fishery recruitment for that year. However, during winter and spring Arctic frontal passages, when the water is chilled and blown out of the estuaries by north winds (i.e. during a "blue norther"), the immigration of PL is delayed (Wenner et al. 1998, Blanton et al. 1999, Benfield and Downer 2001). These events weaken temporally-dependent models, increase the variability in the rate of PL immigration, and increase the variability in estimated density obtained by sampling.

Brown shrimp larvae grow and develop as plankton in shelf waters of the GOM, and many factors lead to a patchy distribution as the PL migrate towards shore and immigrate through passes into bays. As meroplankton, their distribution is governed by seasonal circulation patterns, shelf gyres, wind-driven coastal and tidal currents (Temple and Fischer 1965, 1967, Temple and Martin 1979), and by wind and temperature controlled upwelling and downwelling (Wenner et

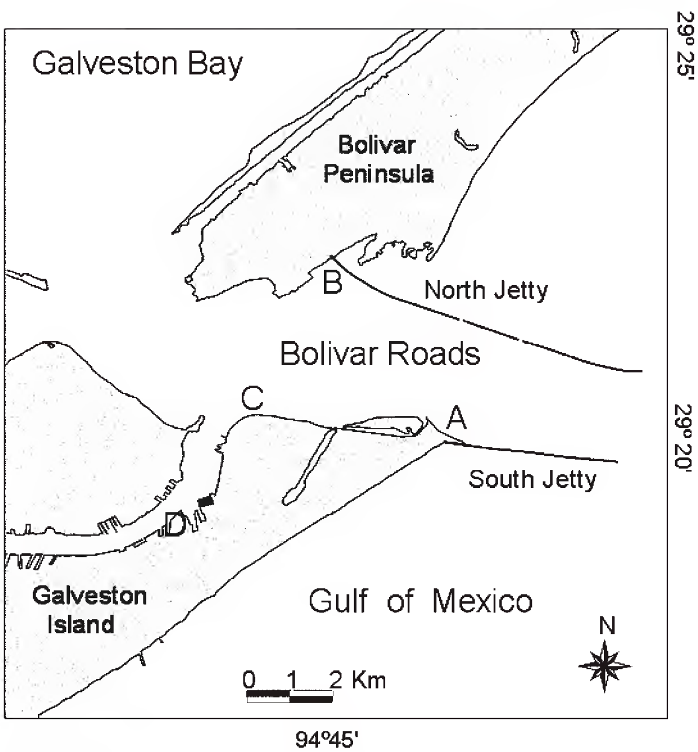

Figure 1.
Postlarval shrimp sampling sites along Bolivar Roads, the main pass connecting Galveston Bay, Texas, with the Gulf of Mexico. Sites: A= South Jetty site, B= North Jetty site, C= Fort Point (USACE water temperature gage), D= Pier 21 (NOS tide gage).

al. 1998, Queiroga et al. 2006). The PL stage is the last of 12 planktonic stages (Cook 1966) that develop in the GOM on a schedule dictated by food availability and environmental conditions, and larvae and PL are transported across the shelf towards shore by coastal currents (Rogers et al. 1993, Rothlisberg et al. 1983, 1995, Criales et al. 2006), and through passes to estuarine nurseries by tidal currents (Lochmann 1990, Herke et al. 1996, Houser and Allen 1996, Criales et al. 2000). Both types of currents can be altered seasonally by winds, particularly in the spring by Arctic frontal passages along the Gulf coast (Smith 1975, 1978). The immigration of PL would be expected to change as these currents change.

The main objective of this paper is to elucidate the potential for drawing erroneous conclusions about the abundance of immigrating PL brown shrimp by looking at time and space differences in PL abundance. Three sets of collections of immigrating PL were examined for variability in a "spring peak" and in short temporal and spatial variability in abundance estimates. Though the accuracy of an abundance estimate cannot be measured because the true number of immigrating PL can never be known, the data presented here demonstrate that monitoring programs also are limited in the precision of their abundance measurements.

Materials and Methods

Sampling Procedures

The studies were conducted at Bolivar Roads (29° 20' N, 93° 44' W), a jettied tidal pass forming the main entrance into Galveston Bay, Texas, from the GOM (Figure 1). The South Jetty site (Figure 1, point A) and the North Jetty site (Figure 1, point B) were located along the south and north shorelines of the pass, respectively. These beach sites were sandy and mostly gently sloping (~ 1:25) with some small bottom ripples that changed weekly due to tidal currents and wave action.

All PL collections were made using a modified Renfro beam trawl constructed with a 1.8 m galvanized iron pipe (12.7 mm) that spread a 1.5 m semi-conical trawl of 1 x 2 mm mesh woven nylon netting (Renfro 1963). During a tow, the net was opened by a floating head rope while the foot rope was kept on the bottom by multiple weights and the pipe beam. A standard tow involved walking the net around a 23 m radius semi-circular path from shore to shore along a central pivot point. Maximum water depth sampled was 1.2 m, and towing speed was about 1 m sec^{-1} . The catch was preserved in 5-10% buffered formalin. Each standard tow swept about 102 m^2 of bottom and filtered about 36 m^3 of water based on water depth, mouth opening, and distance towed. Because the vertical distribution of PL was unknown and the volume of water filtered was only approximated, the number of PL per tow (PL/tow) is used to present catch/abundance data.

Spatial and temporal effects on variability in PL abundance were studied during intensive sampling in spring 1987. Postlarvae were collected during daylight (0800-1700) and nighttime (2000-0400), Monday through Friday from 9 March – 3 April 1987 at both South Jetty and North Jetty collection sites. Collections each week were scheduled to include at least two ebb and two flood tides during daylight and darkness based on predicted tide tables (NOS 1986). For each collection three beam trawl tows were made at each site, tow paths being spread along the shoreline with 25 m between ending point of one tow and starting point of the next. During the third week, separate crews sampled both sites simultaneously, and one hour after the first collection, a second collection was made at the South Jetty site to discover if significant differences should be expected over a 1 hr period—the usual travel time between sites.

The effect of tow length (m) was examined at the South Jetty site on 17 September 1987. Ten sets of tows were made between 0840 and 1200 h. For each set, three semicircular concentric tows using radii of 11, 23, and 46 m (37, 75, and 150 ft) were made simultaneously. Only the 46 m radius tows had to be overlapped slightly because the length of shoreline shallows was limited. Because tows reached from the shore into open shallow water, variation in tow length also incorporated differences in abundance due to water depth.

Hydrographic and weather data were recorded during each collection. Hourly wind speed (Kmph) and direction and air temperature (°C) data were obtained from the National Weather Service for spring 1987. Also, hourly water temperature (°C) data were obtained from the U.S. Army

Corps of Engineer's gauge at Fort Point (Figure 1, point C), and tide levels (cm) were obtained from the National Ocean Service gauge at the Galveston Pier 21 that is located beside the Galveston Channel (Figure 1, point D) for spring 1987.

All PL in each catch were picked, identified, and counted for the normal monitoring samples. Catches were sub-sampled ($\geq$ 12.5% of total catch, for a target of 200 PL) when catches were large in the intensive sampling study. White shrimp, *Litopenaeus setiferus* (Linnaeus) PL were separated by key characteristics including presence or absence of dorsal carinal spines (Williams 1959, Cook 1966, Ringo and Zamora 1968) and by size. In the year-round monitoring samples (1960-1975, 1983-1987, 1989), PL were identified as white, pink (*Farfantepenaeus duorarum* (Burkenroad)), or brown shrimp. In the intensive sampling and the three radii studies PL were identified as white or brown shrimp. Any potential pink shrimp PL were pooled with brown shrimp. It is likely that over 95% of the pink and brown (grooved) PL in this research were brown shrimp based on key characteristics, season of occurrence, and the species composition of the shrimp fisheries in Galveston Bay (Baxter et al. 1988). Studies to separate grooved PL and juveniles up to 7 mm carapace length are ongoing because characteristics in published keys appear to be only about 60% accurate for separating pink and brown shrimp specimens collected in the northwestern GOM estuaries (J. Ditty, pers. comm., NMFS, Galveston, TX).

Analysis

Regression and correlation analyses between PL abundance and environmental conditions were estimated using MS Excel 2000, Sokal and Rohlf (1969), and SAS (1987) for personal computers. Postlarval abundance was transformed using Ln(PL+1) to reduce the variance-to-mean correlation (Berry and Baxter 1969, Caillouet et al. 1970); however, an F_{max} test revealed the variances were still heteroscedastic. Thus, Wilcoxon matched-pairs signed-ranks tests (Siegel 1956) and graphical inspections were used to compare PL abundance from tow to tow, hour to hour, day to night, day to day, site to site, and among radii.

The abundance data from earliest monitoring of PL immigration covered 1960-1966 (Baxter and Renfro 1966, Berry and Baxter 1969) and has been combined here with additional data collected during 1967-1975, 1983-1987, and 1989. Early collections usually did not include replicates at a site, so a daily datum for a year was from either a single sample or from the geometric mean of single samples from the South and North Jetty sites. In the 1980's triplicate samples were taken twice per week at the South Jetty site. Daily means for all years combined were calculated using daily data or means for as many years as were sampled for that Julian day. Multiple moving averages (MA) were calculated, including from 2 to 6 d. Each MA included one or more days leading up to and including the day of record; the more days included, the smoother the spring peak. A quadratic regression analysis was used to determine the spring peak in PL abundance because it yields a parabolic shape with a peak and appeared to have the best potential fit to the data when viewed in a scatter plot. Day, the independent variable, was the Julian day of the year and ranged from 1 (1 January) to 161 (10 June).

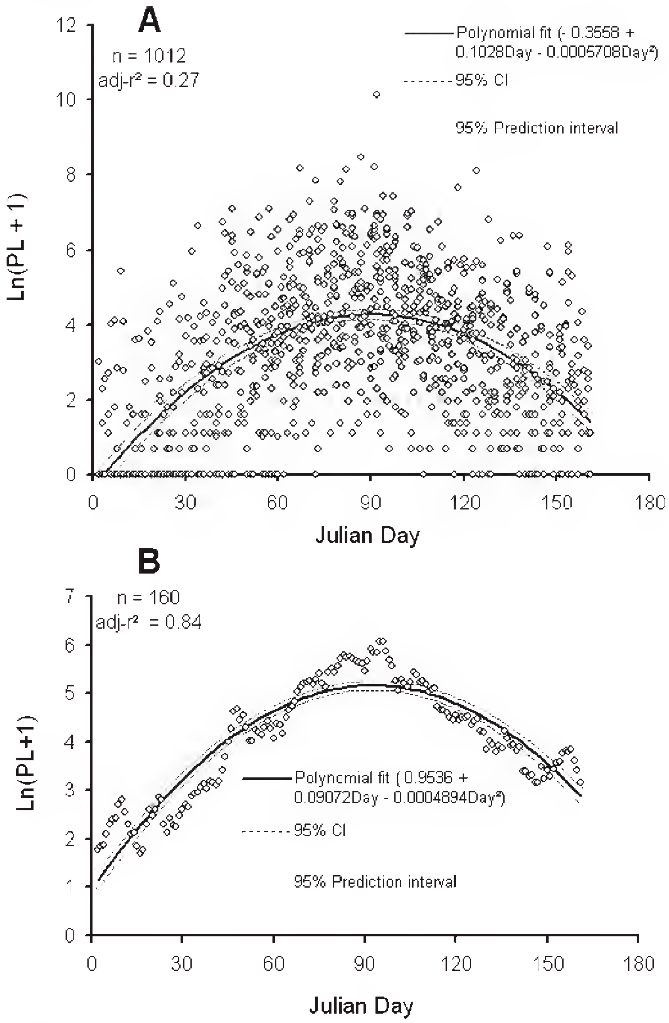

Figure 2.
Postlarval brown shrimp sampled by NMFS monitoring of Bolivar Roads, Texas.

(A) Daily mean catches for each year for January through mid-June of 1960-1975, 1983-1987, and 1989.

(B) The quadratic regression line for 6-day moving average of the daily mean abundance of years combined. 95% confidence limits (dotted lines) and 95% prediction intervals (dashed lines) are indicated in each section. Julian Day 1 – January 1; Julian Day 150 = May 30.

Relationships between PL abundance and water temperature (°C), salinity (‰), and north-south wind vectors (see below) were examined graphically and by correlation analyses. North-south wind vectors were calculated using wind speeds and directions. Northwest, north, and northeast directions produced negative vectors, east and west produced zero vectors, and southeast, south and southwest directions produced positive speed vectors (Kmph) for correlation analyses.

Results

Postlarval Brown Shrimp Spring Peak

An inspection of 22 yr of January through early June abundance data from collections in Bolivar Roads revealed that PL brown shrimp immigrated into Galveston Bay throughout the year. Immigration was found even during the coldest months, but was usually greatest during March and April (Figure 2). High abundances (> 1000 PL/tow) were found February through May depending on the year (Figure 2A). Using Ln(PL+1)-transformed daily mean catches during each year a quadratic regression produced an adjusted-r^2 of only 0.27 (n = 1020). The quadratic regression using 6-d moving averages (MA6) of daily means for years combined formed an acceptable spring peak that accounted for about 84% of the variability (Figure 2B) and established the peak on 2 April from the equation: MA6 $\mathrm{Ln(PL+1)} = 0.8736 + 0.09037\mathrm{Day} - 0.0004934\mathrm{Day}^2$ (adjusted-r^2 = 0.84, n = 159, $p < 0.001$). The mean and 95% confidence limits for Ln(PL+1)-transformed abundance data for the 62 samples (all years) collected during 30 March – 5 April, the week of the peak, were 144 PL/tow and 88-235 PL/tow, respectively, compared to the regression peak of 149 PL/tow and 95% confidence limits of 57-392 PL/tow.

1987 Intensive Spring Sampling Study

Abundance of brown shrimp PL ranged from 0 to 24,616 PL/tow, with a mean of 409 PL/tow (se = 119, n = 262 samples; Table 1) during spring 1987. No white shrimp PL occurred in the samples; they were never found before May during 22 yr of monitoring in Bolivar Roads. This maximum catch (24,616) was higher than any recorded catch during the 22 yr of standard monitoring. The means for triplicate tows ranged from 0.7 to 15,673 PL/tow and averaged 440 (se = 207, n = 77). Means for a calendar day (n = 12; 3 day and 3 night at the two sites) ranged from 18 to 4,488 PL/tow with the grand daily mean being 426 (se = 218, n = 20). The means for the North and South Jetty sites were 82 (se = 14, n = 57) and 962 PL/tow (se = 523, n = 55) for daylight, 288 (se = 42, n = 60) and 449 PL/tow (se = 110, n = 60) for night, and 188 (se = 24, n = 117) and 694 PL/tow (se = 257, n = 115) overall, respectively. For all daytime and nighttime tows the means were 514 (se = 259, n = 112) and 369 PL/tow (se = 59, n = 120), respectively. High variability in abundance found among the triplicates, day/night, dates, and sites was not constant and may not have been obvious without intense sampling (Figure 3).

Times and Sites

Observing changes in PL over various periods is useful for understanding PL influxes through passes and for establishing sampling regimes. The largest coefficient of variation (CV) for triplicate Ln(PL+1)-transformed abundance was 86.6%, and the smallest was 0.6%; both were for daytime collections at the South Jetty. Abundance in nighttime triplicates generally varied less than those in daytime triplicates (Table 2). Among all triplicate samples, 55% had CV's ≤ 10% and 77% had CV's ≤ 20%.

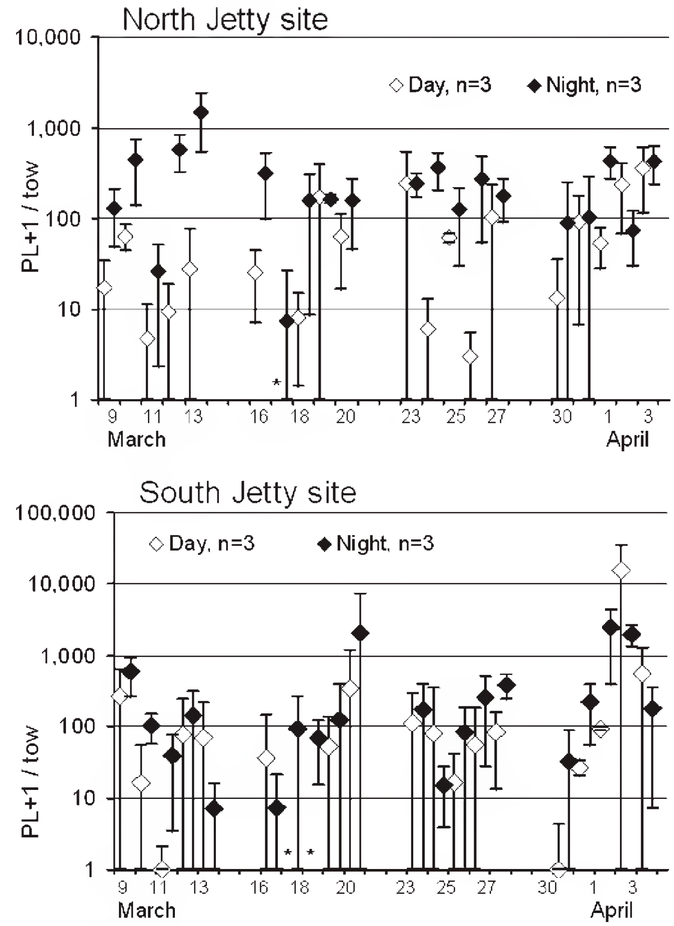

Figure 3.
Means and 95% confidence intervals for abundance of postlarval brown shrimp during March 1987 in Bolivar Roads, Texas for two sites. Sampling occurred Monday through Friday for 4 weeks. Means are for triplicate tows using Renfro beam trawls.
** = no data.*

Postlarval abundance varied from hour to hour, and had a mean absolute difference of 104 PL/tow (se = 64) for the ten paired sets of triplicate samples. This difference was less than the mean of the 20 triplicate means, 154 PL/tow (se = 40), used for the comparison. Changes occurring during half a day (~12 h) were confounded by the light factor—day becoming night and vice versa. Night abundance at each site was greater than the corresponding day 78% of the time (Table 1 and Figure 3). The mean absolute difference over 12 h was 650 PL/tow (se = 286, n = 66). This difference was considerably larger than the mean, 446 PL/tow (se = 212), of the 74 triplicate means used for the comparisons. Changes in abundance from day to day (~24 h) were tested by comparing means from one daytime sampling to the next and from one nighttime sampling to the next for each site separately. The mean absolute difference was 767 PL/tow (se = 363, n = 59). This difference was also considerably larger than the mean, 451 PL/

TABLE 1. *Brown shrimp postlarval abundance in Bolivar Roads, Texas during spring 1987, as caught in Renfro beam trawl shoreline based tows. Each tow swept 102m² of bottom and filtered about 36m³ of water; nd = no data; +1 Hr = samples taken one hour later at same sites.*

	DAY						**NIGHT**						
	North Jetty Site			South Jetty Site			North Jetty Site			South Jetty Site			Daily
Date (1987)	Tow 1	Tow 2	Tow 3	Tow 1	Tow 2	Tow 3	Tow 1	Tow 2	Tow 3	Tow 1	Tow 2	Tow 3	mean
9-Mar	24	11	16	127	253	420	99	163	127	721	458	583	250.2
10-Mar	36	56	63	17	10	6	469	311	554	103	84	122	152.6
11-Mar	7	5	2	1	0	1	16	27	35	31	56	31	17.7
12-Mar	12	11	5	95	6	135	554	686	489	86	126	216	201.8
13-Mar	21	13	49	23	138	49	1,672	1,040	1,703	9	9	3	394.1
16-Mar	28	31	17	22	3	86	322	396	222	13	7	2	95.8
17-Mar	nd	nd	nd	nd	nd	nd	16	4	2	35	167	78	50.3
18-Mar	5	9	10	197	nd	nd	201	89	182	58	94	55	90.0
19-Mar	73	250	191	16	82	57	173	160	160	248	76	48	127.8
20-Mar	52	53	85	729	68	254	153	205	115	4,503	922	746	657.1
23-Mar	219	135	376	173	37	126	255	267	213	141	270	115	193.9
+ 1 Hr				59	20	45				154	215	175	111.3
24-Mar	8	3	7	3	35	202	434	310	344	18	10	18	116.0
+ 1 Hr				20	93	50				128	118	119	88.0
25-Mar	59	59	64	7	17	26	156	135	82	58	66	129	71.5
+ 1 Hr				23	19	34				891	912	473	392.0
26-Mar	4	2	3	70	3	99	346	175	287	162	265	348	147.0
+ 1 Hr				2	1	10				124	304	220	110.2
27-Mar	80	91	57	109	53	90	145	175	218	345	363	453	181.6
+ 1 Hr				87	36	64				528	365	155	205.8
30-Mar	3	20	16	2	2	0	71	38	161	37	52	8	34.2
31-Mar	60	85	127	24	29	27	24	123	164	286	227	152	110.7
1-Apr	42	62	55	95	90	92	512	404	390	3,194	1,558	2,534	752.3
2-Apr	287	265	159	12,644	24,616	9,760	61	94	66	1,776	1,856	2,272	4,488.0
3-Apr	457	355	260	237	589	830	384	387	522	259	130	150	380.0

tow (se = 212), of the 75 triplicate means used for the comparisons. The increase in absolute differences with increasing time between collections indicates that the abundance of PL moving through the pass is extremely dynamic and that short term, even hourly, changes could be substantial.

Differences in abundance between north and south jetty sites were examined by comparing means of triplicate catches for days and nights separately. The mean absolute difference was 730 PL/tow (se = 418, n = 37), and is considerably larger than the grand mean, 455 PL/tow (se = 215), of the 74 triplicates used in the comparisons. This difference was very close to that found for changes that occurred over about 24 h, and larger than that found over 12 h.

Sources of variation in PL abundance were ranked according to magnitude of CV. CV's were calculated for abundance based on replicates (triplicates), hour to hour, day-night, sites, and dates. The CV was highest for sites, followed by dates, day-night, replicates, and hours, respectively (Table 3). However, when abundance was Ln(PL+1)-transformed, the hierarchy of CV's changed to dates and day-night being greatest, followed closely by sites, and then hours and finally replicates.

Tides and Environment

Weather and tides varied considerably, as is typical for spring along the northern GOM (Figure 4A). Water temperature (Figure 4B) ranged from 8-28°C and salinity from 15-28‰ at the sampling sites. Pearson product moment correlations between Ln(PL+1)-transformed abundance and water temperature ($r = 0.22$, $p = 0.057$, $n = 77$), salinity ($r = 0.08$, $p = 0.464$, $n = 77$), and wind speed vectors ($r = -0.07$, $p = 0.527$, $n = 77$) were weak and not significant. Postlarval abundance was depressed during two significant "blue northers", one on 10-11 March and a larger one on 29

TABLE 2. *Frequency distributions of coefficients of variation of postlarval brown shrimp caught in triplicate samples. Collections were made along Bolivar Roads, Texas from 9 March – 3 April 1987. Catches had been transformed using: Ln(PL+1). PL = postlarvae.*

	NORTH JETTY		SOUTH JETTY			
CV (%)	**Day**	**Night**	**Day**	**Night**	**Sum**	**Cum. %**
0 - 10	9	16	6	17	48	55
11 - 20	7	2	5	5	19	77
21 - 30	2	1	4	2	9	87
31 - 40	1		1	1	3	90
41 - 50		1	1		2	93
51 - 60			3		3	96
61 - 70			1		1	97
71 - 80						97
81 - 90			2		2	100
90 - 100						100
n =	19	20	23	25	87	

March, and rebounded as each "norther" abated (Figure 5).

Flood-tides, which bring the PL into the pass from the Gulf waters, did not appear to be particularly important when weekly PL catches were examined with respect to tides, day-night, and location (Figure 6). In only 5 of 8 North Jetty cases and only 2 of 6 South Jetty cases were PL abundances greater on flood tides than on ebb tides. Eddy currents located between the ship channel and the shoreline probably added to the disconnect between abundance and tidal flows.

The Effect of Tow Radius

Postlarval abundances for the ten replicates of each radius differed by nearly two orders of magnitude along this short, 500 m, stretch of beach (Table 4). Among the standard 23 m radius tows, grooved shrimp, white shrimp, and total shrimp catch ranges were 48 to 3,224, 9 to 1,478, and 57 to 4,702 PL/tow, respectively. These large differences for both species were from just 360 m along the beach (Tows 1 and 8).

Short tow (11 m) abundances did not correlate well with those in the standard tows ($r = -0.18$, $p = 0.61$, $n = 10$), and when doubled to match the tow length of the standard, they were always less than standard tow abundances. Total PL abundances from the standard and long tows (46 m) correlated well ($r = 0.88$, $p = 0.002$, $n = 9$), but when standardized for tow length, the standard tow catches were greater than those of the long tows 78% of the time. Such results indicate the PL were irregularly distributed out from shore as well as along shore, with more PL appearing to be in the intermediate depth that was sampled most by the standard radius.

Discussion

Federal and state fishery biologists and managers in the GOM have been particularly interested in maintaining the valuable brown, white, and pink shrimp fisheries. While other fisheries have been or are being over-fished and harvests declining, the shrimp harvests are holding fairly steady or declining only slightly through 2006 (NMFS 2007). An important correlation linking the adult shrimp harvest from the GOM with estuarine marsh nursery habitat (Turner 1977) coupled with the increase in man's developments along the bay shores suggests dismantled or degraded salt marsh nursery habitat may lead to reductions in shrimp harvests. For example, Browder et al. (1989) pointed to the insidious correlation between marsh break-up and shrimp populations, in that shrimp production increases as break-up increases to a point beyond which both crash. Marsh restoration efforts are not keeping pace with marsh destruction, and another few decades of marsh destruction could well lead to significant decreases in shrimp populations and harvests in the GOM.

The objectives of monitoring PL brown shrimp immigration are to better understand this shrimp's annual cycle, and then to use the intensity and/or timing of spring estuarine immigration of PL to forecast the summer harvests. High densities of immigrating brown shrimp PL have been not-

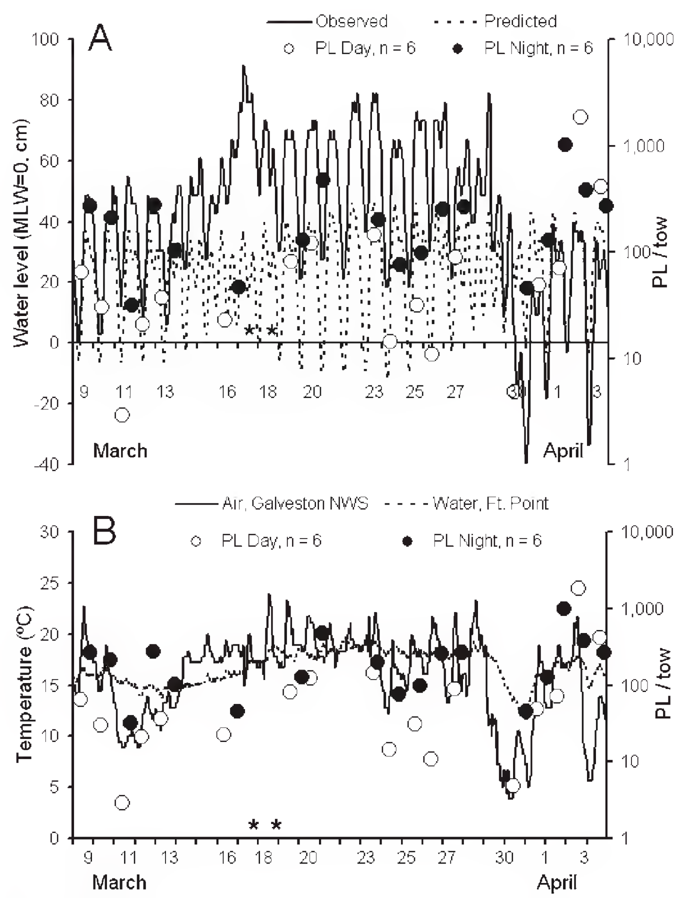

Figure 4.
Comparison of postlarval abundance from an intensive sampling study in Bolivar Roads, just east of Galveston, Texas in 1987 with environmental variables.

A. Hourly observed and predicted water heights (NOS).

*B. Hourly air temperatures (NWS) and water temperatures (USACE). * = no data.*

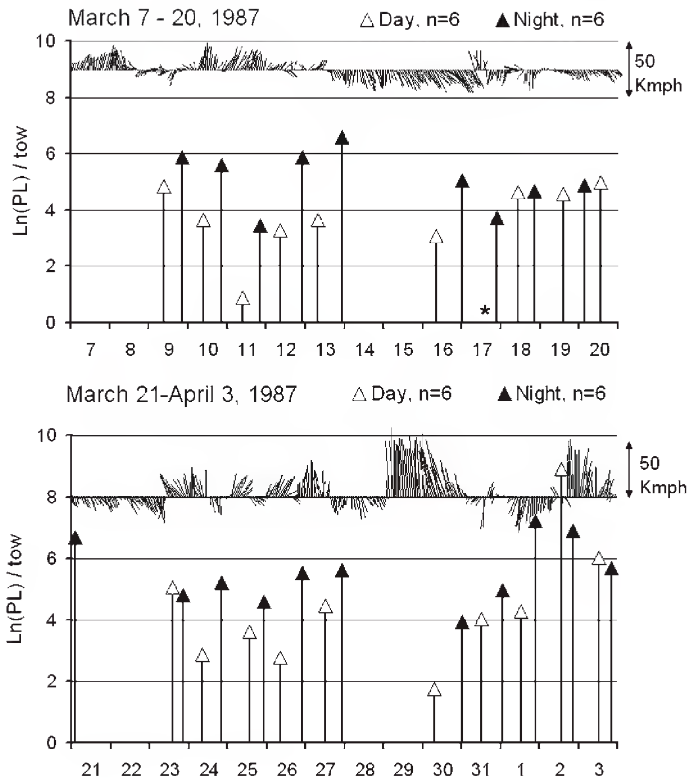

Figure 5.
*Hourly wind speed (Kmph) vectors are compared with postlarval brown shrimp abundance (vertical arrows) from Bolivar Roads, Texas, Spring 1987. A vertical wind vector above the horizontal axis indicates a wind from the north, and vector length may be compared with the 50 Kmph double headed arrow on the right. * = no data.*

ed during March and April in Texas (Copeland and Truitt 1966, Berry and Baxter 1969, Kutkuhn et al. 1969), Louisiana (Caillouet et al. 1971, Rogers and Herke 1985, Rogers et al. 1993), North Carolina (Williams 1964, Williams and Deubler 1968), and South Carolina (DeLancey et al. 1994). The timing of peak abundance has differed substantially from year to year, and some years the peak was absent—exchanged for intermittent highs and lows. These variations offered potential annual differences for forecasting models.

Without data from multi-year monitoring, characterization of the "spring peak" lacks substantive form. Individually, many of the previously cited studies suggested a "spring peak" but were unable to define it. Fortunately, annual sampling by Baxter allowed calculation of a regression equation to define the peak abundance which may be valid for the Texas coast. A similar regression based on long term data may also define peaks and migrations for *F. aztecus* PL along the north central GOM and Carolina coasts. However, one should not expect to reliably find large numbers of PL in a pass on a date based on the regression because many environmental and biological factors operate on PL distributions to reduce or inflate numbers on any particular day of a particular year. At present, the importance of the spring peak seems to be that it conceptualizes the importance of the estuarine habitat during that time of year for perpetuating brown shrimp. To use its changes in magnitude and/or timing of occurrence as forecasting variables will depend on our ability to adequately assess and evaluate the changes, and that will require addressing short-term variability in PL density measurements.

Small scale variability in density estimates appears high and has a large range that is significant over time and space. Thus, this variability can cause the annual influx event to be misrepresented in small-scale sampling efforts. For Bolivar Roads this study reported a maximum of 24,616 PL/tow or 684 PL/m^3 whereas Baxter and Renfro (1966) reported a maximum of 131 PL/m^3 and Duronslet et al. (1972) reported a mean high of barely over 1 PL/m^3. Arnold et al. (1960) observed in the same area that PL "... were swimming at the surface and so concentrated that several thousand could be caught with a single scoop of a dip net. On each occasion, large numbers of fish (mostly pinfish and anchovies) could be seen decimating the relatively helpless shrimp." These varying reports suggest high density collections may be quite ephemeral and no more important than some intermediate density for distributing PL in the bay. Other maxima of note in Texas are: 76 PL/m^3 along the front beach of Galveston Island during the spring (Benfield and Downer 2001), 60 PL/m^3 at Rollover Pass, Texas, from plankton tows (Berry and Baxter 1969), and 299 PL/m^3 in plankton net collections in Cedar Bayou that connects the GOM to Mesquite Bay (King 1971).

The greatest abundance reported here, and the largest in 22 yr of sampling, was 684 PL/m^3 and occurred on the theoretical spring peak and just three days after a strong "blue norther" had blown through and reduced PL density to < 1 PL/m^3. Similar increases in PL after northers have been reported in Louisiana (Rogers et al. 1993). The norther not only pushed the water out of the bay and held it out for about a day, but also chilled the shallow water to below 10 °C which probably caused PL to bury themselves in the bottom (Aldrich et al. 1968). Postlarvae may also have

TABLE 3. *Sources of variation in postlarval (PL) brown shrimp catches that used Renfro beam-trawls to sample at shoreline sites in Bolivar Roads, Texas. Coefficient of variation (CV) indicates the importance of the factor in contributing to the total variance.*

Factor	n	$\bar{x}$	Variance	CV	CV of Ln(PL+1)-transformed catches
Triplicates	87	440	881,636	214	14
Hourly	10	154	23,915	100	21
Day/Night	66	472	2,869,345	359	32
Site	39	439	3,241,015	410	30
Date	18	465	3,381,709	375	32

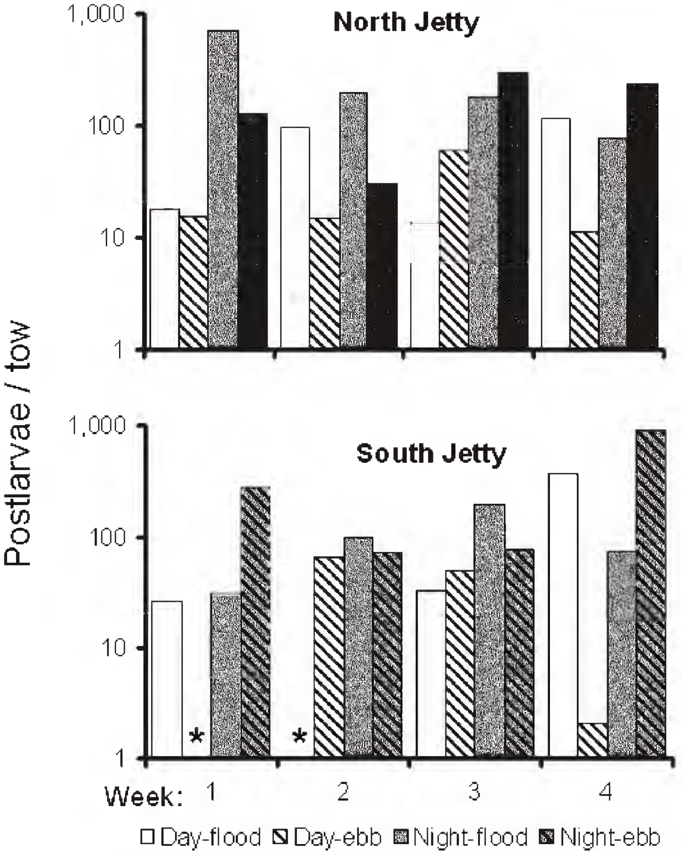

Figure 6.
*Weekly mean abundance of postlarval brown shrimp showing differences due to tidal flows, sites, and day or night conditions at two sites in Bolivar Roads, Texas, Spring 1987. * = no data.*

been concentrated in the near-offshore area by the offshore winds of the norther. Smith (1975, 1978) showed that as cold winds blow offshore they carry surface water offshore and consequently bring subsurface water towards the coast. The cold surface water may also have made PL drop into the warmer mid- and bottom water, concentrating them there, and bringing them towards shore. With the return of warmer onshore winds from the southeast and the rising water flooding back into the emptied bay, the PL are then carried into the bay. The observed super-abundance may have resulted from the addition of PL that emerged from the bottom to join those concentrated near shore by wind and cold and those approaching the coast in the normal manner. Knott et al. (1994) found wind forcing to be important for white shrimp PL and blue crab ingress to South Carolina passes.

It also appears that it might take more than one tidal flood to transport the accumulated PL through Bolivar Roads, a large pass with eddies along its sides. This could add more PL to the emerging group, if they had been trapped in the shallows during the norther as they immigrated. The fact that my data are from shoreline sampling may explain some of the lack of correlation between abundances and environmental and tidal conditions. By the time PL reach the sides of the pass where they were sampled tidal conditions may have changed, and their immigration slowed by slower currents and more eddies. Thus, the abundances observed may represent an accumulation rather than an instantaneous occurrence which would be reflective of environmental conditions when they initially arrived.

Although the existence of the variability in abundance during the spring offers potential for forecasting the shrimp fishery, the numerous sources causing differences in abundance estimates appear not to have been accommodated in past monitoring regimes. For example, the currently noncorrelative existence between environmental factors and PL abundance is not a surprise as brown shrimp PL are widely tolerant of temperature and salinity (Zein-Eldin and Aldrich 1965), but it will complicate selection of relationships for forecasting models, and will diminish the usefulness of PL abundance for forecasting unless a connection can be found. Brown shrimp PL immigration continues through the summer with another smaller peak occurring in the fall, all of which offer additional potential for population modeling. A strong sampling regime will be required to address and separate the combination of biological and environmental factors that are responsible for changes in fishery harvest later in the year. Criales et al. (2006) found a similar need while studying pink shrimp PL immigration to Florida Bay.

High variability in abundance of PL was observed in studies designed to examine effects of time, date, day/night, tide, and tow distance. Some of this variability had been noted previously by Berry and Baxter (1969), Caillouet et al. (1968), Lochmann (1990), and Benfield and Downer (2001). Such extensive variability as was found over short time periods and distances illustrates that collecting only a few samples a couple of times per week or month, and at one or two sites, is likely to be inadequate to describe the dynamic PL immigration in a pass during an expanded time period. Limited data so gathered is potentially misleading, and would not likely be useful in forecasting the fishery harvest as was noted by Benfield and Downer (2001) for shrimp, or for predicting changes in fish populations (Osenberg et al. 1994). To increase the power of a monitoring program for immigrating PL, it seems best to increase sampling to account for the factor contributing the largest variance. Our CV calculations suggest that increasing the number of dates and sites sampled would add most to a sampling regime, with both day and night sampling and replicates having less importance.

This research pertained mainly to *F. aztecus* PL, but these high variability problems in Bolivar Roads likely apply to other estuarine passes as well, and to other species of shrimp, fish, and crab larval and PL populations that immigrate through passes. The strength of PL shrimp immigration may be a good indicator of future shrimp fishery harvest, but obtaining an accurate measurement of immi-

TABLE 4. *Postlarval shrimp catches during the triple radius test at the South Jetty site in Bolivar Roads, Texas, 17 September 1987. Brown = brown shrimp; White = white shrimp; sd = standard deviation; CV = coefficient of variation.*

Radius		11 m			23 m			46 m	
SET	Brown	White	Total	Brown	White	Total	Brown	White	Total
1	24	1	25	48	9	57	182	30	212
2	29	3	32	138	28	166	142	39	181
3	76	8	84	286	51	337	244	56	300
4	95	15	110	293	65	358	289	80	369
5	30	5	35	235	61	296	269	50	319
6	46	18	64	175	72	247	318	103	421
7	90	62	152	604	208	812	1,019	294	1,313
8	27	41	68	3,224	1,478	4,702	1,616	576	2,192
9	7	7	14	2,498	821	3,319	nd	nd	nd
10	89	125	214	400	95	495	742	357	1,099
$\bar{x}$:	**51**	**29**	**80**	**790**	**289**	**1,079**	**536**	**176**	**712**
sd:			**63**			**1,592**			**684**
CV (%):			**79**			**148**			**96**

gration may not be possible. Thus, we may need to also consider environmental parameters that affect juvenile growth and survival to provide an accurate fishery forecast.

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Literature Cited

Aldrich, D. V., C. E. Wood, and K.N. Baxter. 1968. An ecological interpretation of low temperature responses in *Penaeus aztecus* and *P. setiferus* postlarvae. Bulletin of Marine Science 18:61-71.

Arnold, E.L. Jr., R.S. Wheeler, and K.N. Baxter. 1960. Observations on fishes and other biota of East Lagoon, Galveston Island. United States Fish and Wildlife Service Special Scientific Report Number 344, 30 p.

Baxter, K.N. 1963. Abundance of postlarval shrimp - one index of future shrimping success. Proceedings of the Gulf and Caribbean Fisheries Institute 15:61-71.

Baxter, K.N. and W.C. Renfro. 1966. Seasonal occurrence and size distribution of postlarval brown and white shrimp near Galveston, Texas, with notes on species identification. Fishery Bulletin 66:149-158.

Baxter, K.N. and L.F. Sullivan. 1986. Forecasting offshore brown shrimp catch from early life history stages. In: Proceedings, Shrimp Yield Prediction Workshop. Texas A&M Sea Grant Publication, TAMU-SG-86-110, College Station, TX, USA, p. 22-36.

Baxter, K.N., C.H. Furr, and E. Scott. 1988. The commercial bait shrimp fishery in Galveston Bay, Texas, 1959-87. Marine Fisheries Review 50:20-28.

Benfield, M.C. and R.G. Downer. 2001. Spatial and temporal variability in the nearshore distributions of postlarval *Farfantepenaeus aztecus* along Galveston Island, Texas. Estuarine Coastal and Shelf Science 52:445-456.

Berry, R.J. and K.N. Baxter. 1969. Predicting brown shrimp abundance in the northwestern Gulf of Mexico. Food and Agriculture Organization of the United Nations Fisheries Report 57:775-798.

Blanton, J.O., F.E. Werner, A. Kapolnai, B.O. Blanton, D.

Knott, and E.L. Wenner. 1999. Wind-generated transport of fictitious passive larvae into shallow tidal estuaries. Fisheries and Oceanography 8:210-223.

Browder, J.A., L.N. May, A. Rosenthal, J.G. Gosselink, and R.H. Baumann. 1989. Modeling future trends in wetland loss and brown shrimp production in Louisiana using thematic mapper imagery. Remote Sensing of the Environment 28:45-59.

Caillouet, C.W., Jr., B.J. Fontenot, and R.J. Dugas. 1968. Diel fluctuations in catch of postlarval white shrimp, *Penaeus setiferus* (Linnaeus), with the Renfro beam trawl. Bulletin of Marine Science 18:829-835.

Caillouet, C.W., Jr., W.S. Perret, and R.J. Dugas. 1970. Diel fluctuations in catch of postlarval brown shrimp, *Penaeus aztecus* Ives, with the Renfro beam trawl. Bulletin of Marine Science 20:721-730.

Caillouet, C.W., Jr., B.J. Fontenot, W.S. Perret, R.J. Dugas, and H.F. Herbert. 1971. Catches of postlarval white shrimp, *Penaeus setiferus* (Linn.), and brown shrimp, *P. aztecus* Ives, and temperature and salinity observations in Vermilion Bay, Louisiana, March 1963 to April 1967. Data Report 64, Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Washington, D.C. USA, 42 p.

Cook, H.L. 1966. A generic key to the protozoea, mysis, and postlarval stages of the littoral Penaeidae of the northwestern Gulf of Mexico. Fishery Bulletin 65:437-447.

Copeland, B.J. and M.V. Truitt. 1966. Fauna of the Aransas Pass Inlet, Texas. II. Penaeid shrimp postlarvae. Texas Journal of Science 18:65-74.

Criales, M.M., M.J. Bello, and C. Yeung. 2000. Diversity and recruitment of penaeoid shrimps (Crustacea: Decapoda) at Bear Cut, Biscayne Bay, Florida, USA. Bulletin of Marine Science 67:773-788.

Criales, M.M., J.D. Wang, J.A. Browder, M.B. Robblee, T.L. Jackson, and C. Hittle. 2006. Variability in supply and cross-shelf transport of pink shrimp (*Farfantepenaeus duorarum*) postlarvae into western Florida Bay. Fishery Bulletin 104:60-74.

Darnell, R.M., R.E. Defenbaugh, and D. Moore. 1983. Northwestern Gulf shelf bio-atlas; a study of the distribution of demersal fishes and penaeid shrimp of soft bottoms of the continental shelf from the Rio Grande to the Mississippi River Delta. Open File Report Number 82-04, Minerals Management Service, Gulf of Mexico Outer Continental Shelf Regional Office, Metairie, LA, USA, 438 p.

DeLancey, L.B., J.E. Jenkins, and J.D. Whitaker. 1994. Results of long-term, seasonal sampling for *Penaeus* postlarvae at Breach Inlet, South Carolina. Fishery Bulletin 92:633-640.

Duronslet, M.J., J.M. Lyon, and F. Marullo. 1972. Vertical distribution of postlarval brown, *Penaeus aztecus*, and white, *P. setiferus*, shrimp during immigration through a tidal pass. Transactions of the American Fisheries Society 101:748-752.

Herke, W.H., B.D. Rogers, V.L. Wright, and W.H. Bradshaw. 1996. Postlarval *Penaeus aztecus* and *P. setiferus* transport into, and distribution within, adjacent weired and unweired ponds. Wetlands 16:197-207.

Houser, D.S. and D.M. Allen. 1996. Zooplankton dynamics in an intertidal salt-marsh basin. Estuaries 19:659-673.

King, B.D. III. 1971. Study of migratory patterns of fish and shellfish through a natural pass. Technical Series 9, Texas Parks and Wildlife Department, Austin, TX, USA, 54 p.

Klima, E. F., P.F. Sheridan, K. N. Baxter, and F. J. Patella. 1986. Review of the 1985 Texas closure for the shrimp fishery off Texas and Louisiana. Technical Memorandum, NMFS-SEFC-173, Department of Commerce, National Oceanic and Atmospheric Administration, Washington, D.C., USA, 72 p.

Knott, D.M., E.L. Wenner, C.A. Barans, and B.W. Stender. 1994. The influence of wind forcing on ingress of postlarval white shrimp and blue crab into the North Edisto Inlet, SC. Proceedings, 24th Annual Benthic Ecology Meeting, Columbia, SC, USA, p. 97 (abstract).

Kutkuhn, J.H., H.L. Cook, and K.N. Baxter. 1969. Distribution and density of prejuvenile *Penaeus* shrimp in Galveston entrance and the nearby Gulf of Mexico (Texas). Food and Agriculture Organization of the United Nations, Fisheries Report 57:1075-1099.

Lochmann, S. E. 1990. Mechanisms of transport of estuarine-related larval fish and invertebrates through tidal passes of the Texas Coast. Ph.D. thesis, Texas A&M University, College Station, TX, USA, 311 p.

Minello, T.J., R.J. Zimmerman, and E.X. Martinez. 1989. Mortality of young brown shrimp *Penaeus aztecus* in estuarine nurseries. Transactions of the American Fisheries Society 118:693-708.

NMFS (National Marine Fisheries Service). 2007. Fisheries of the United States, 2006. Office of Science and Technology, Fisheries Statistics and Economics Division. Available at: http://www.st.nmfs.noaa.gov/st1/commercial/index.html.

NOS (National Ocean Service). 1986. Tide tables 1987, high and low water predictions. East Coast of North and South America, Including Greenland. Department of Commerce, National Oceanic and Atmospheric Administration, National Ocean Survey, Washington, D.C., USA, 289 p.

Neal, R.A., H.A. Brusher, and L.F. Sullivan. 1983. A survey of brown shrimp resources in the Northwestern Gulf of Mexico. Technical Memorandum, NMFS-SEFC-114, Department of Commerce, National Oceanic and Atmospheric Administration, Washington, D.C., USA, 30 p.

Osenberg, C.W., R.J. Schmitt, S.J. Holbrook, K.E. Abu-Saba, and A.R. Flegal. 1994. Detection of environmental impacts: natural variability, effect size, and power analysis. Ecological Applications 4:16-30.

Queiroga, H., M.J. Almaida, T. Alpuim, A.A.V. Flores, S. Francisco, I.Gonzalez-Gordillo, A.I. Miranda, I. Silva, and J. Paula. 2006. Tide and wind control of megalopal supply to estuarine crab populations on the Portuguese west coast. Marine Ecology Progress Series 307:21-36.

Renfro, W.C. 1963. Small beam net for sampling postlarval shrimp. Circular of the United Sates Bureau of Commercial

Fisheries 161:86-87.

Ringo, R.D. and G. Zamora Jr. 1968. A penaeid postlarval character of taxonomic value. Bulletin of Marine Science 18:471-476.

Rogers, B.D. and W.H. Herke. 1985. Temporal patterns and size characteristics of migrating juvenile fishes and crustaceans in a Louisiana marsh. Research Reports 5, Louisiana State University, Agriculture Center, Baton Rouge, LA, USA, 81p.

Rogers, B.D., R.F. Shaw, W.H. Herke, and R.H. Blanchet. 1993. Recruitment of postlarval and juvenile brown shrimp (*Penaeus aztecus* Ives) from offshore to estuarine waters of the northwestern Gulf of Mexico. Estuarine Coastal and Shelf Science 36:377-394.

Rothlisberg, P.C., J.A. Church, and A.M.G. Forbes. 1983. Modeling the advection of vertically migrating shrimp larvae. Journal of Marine Research 41:511-538.

Rothlisberg, P.C., J.A. Church, and C.B. Fandry. 1995. A mechanism for near-shore concentration and estuarine recruitment of post-larval *Penaeus plebejus* Hess (Decapoda, Penaeidae). Estuarine Coastal and Shelf Science 40:115-138.

SAS Institute Inc. 1987. SAS/STAT Guide for Personal Computers, Version 6 Edition. Cary, NC, USA, 1028p.

Siegel, S. 1956. Nonparametric Statistics of Behavioral Sciences. McGraw-Hill Book Company, New York, NY, USA, 312p.

Smith, N.P. 1975. Seasonal variations in nearshore circulation in the northwestern Gulf of Mexico. Contributions in Marine Science 19:49-65.

Smith, N.P. 1978. Long-period, estuarine-shelf exchanges in response to meteorological forcing. In: J.C.J. Nihoul, ed. Hydrodynamics of Estuaries and Fjords. Elsevier Scientific Publications Company, Amsterdam, Holland, p.147-159.

Sokal, R.R. and F.J. Rohlf. 1969. Biometry. W.H. Freeman and Company, San Francisco, CA, USA, 776p.

Sutter, F.C., III and J.Y. Christmas. 1982. Multilinear models for the prediction of brown shrimp harvest in Mississippi waters. Gulf Research Reports 7:205-210.

Temple, R.F. and C.C. Fischer. 1965. Vertical distribution of the planktonic stages of penaeid shrimp. Publications of the Institute of Marine Science, University of Texas 10:59-67.

Temple, R.F. and C.C. Fischer. 1967. Seasonal distribution and relative abundance of planktonic stage shrimp (*Penaeus* spp.) in the northwestern Gulf of Mexico, 1961. Fishery Bulletin 66:323-334.

Temple, R.F. and J.A. Martin. 1979. Surface circulation in the northwestern Gulf of Mexico as deduced from drift bottles. Technical Report NMFS SSRF-730, United States Department of Commerce, National Oceanic and Atmospheric Administration, Washington, D.C., USA, 13 p.

Turner, R.E. 1977. Intertidal vegetation and commercial yields of penaeid shrimp. Transactions of the American Fisheries Society 106:411-416.

Wenner, E., D. Knott, J. Blanton, C. Barans, and J. Amft. 1998. Roles of tidal and wind-generated currents in transporting white shrimp (*Penaeus setiferus*) postlarvae through a South Carolina (USA) inlet. Journal of Plankton Research 20:2333-2356.

Williams, A.B. 1959. Spotted and brown shrimp postlarvae (*Penaeus*) in North Carolina. Bulletin of Marine Science of the Gulf and Caribbean 9:281-290.

Williams, A.B. 1964. A postlarval shrimp survey in North Carolina. Special Scientific Report Number 3, North Carolina Department of Conservation and Development, Division of Commercial Fisheries, Raleigh, NC, USA, 5p.

Williams, A.B. and E.E. Deubler. 1968. A ten-year study of meroplankton in North Carolina estuaries: assessment of environmental factors and sampling success among bothid flounders and penaeid shrimps. Chesapeake Science 9:27-41.

Zein-Eldin, Z.P. and D.V. Aldrich. 1965. Growth and survival of postlarval *Penaeus aztecus* under controlled conditions of temperature and salinity. Biological Bulletin 129:199-216.

Zimmerman, R.J and T.J. Minello. 1984. Densities of *Penaeus aztecus, P. setiferus* and other natant macrofauna in a Texas salt marsh. Estuaries 7:421-433.

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Preliminary Survey of Fish Community Composition in Seagrass Habitat in Two Back-Reef Lagoons of the Southern Mexican Caribbean

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PRELIMINARY SURVEY OF FISH COMMUNITY COMPOSITION IN SEAGRASS HABITAT IN TWO BACK-REEF LAGOONS OF THE SOUTHERN MEXICAN CARIBBEAN

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ABSTRACT: Little is known about seagrass fish communities in the southern Mexican Caribbean. Diurnal and nocturnal fish community structure in seagrass habitat were compared between back-reef lagoons using a visual census technique in a natural protected area within a national park (Xcalak) and an unprotected area (Mahahual). Seagrass fish communities differed significantly between the two locations in the daytime and Xcalak supported greater total fish densities. Species richness did not differ statistically between locations. Observed nighttime fish communities were characterized by low species richness and low fish abundance when compared to diurnal communities. Heavy tourist use and coastal development may have degraded seagrass habitat at Mahahual causing lower fish abundance. Also, proximity of seagrass to mangrove habitat in Xcalak may have led to increased abundance and differences in species composition between locations. More extensive analysis and monitoring of the relative functioning of back-reef habitats in these two systems is needed as coastal development and fishing pressure continue to threaten the area.

RESUMEN: No se conoce mucho sobre la comunidad de peces en pastos marinos en el sur del Caribe mexicano. La estructura de las comunidades de peces nocturnas y diurnas en pastos marinos se obtuvo mediante censos visuales y se comparó entre la laguna arrecifial de un área protegida (Parque Nacional Arrecifes de Xcalak) y un área no-protegida (Mahahual). Las comunidades de peces fueron diferentes significantemente entre los dos sitios durante el día, Xcalak registró las mayores densidades de peces. No existe diferencia estadísticamente significativa con respecto a la riqueza de especies entre sitios. Las comunidades de peces nocturnas presentaron valores bajos de riqueza de especies y de abundancia con respecto a las comunidades diurnas. El desarrollo turístico y costero de Mahahual, podrían estar degradando el hábitat de pastos marinos, y como consecuencia el registro de bajas abundancia de peces. En contraste, en Xcalak, la proximidad del ecosistema de manglar adyacente a los pastos marinos podría estar influenciando con una mayor abundancia de peces y cambios en la composición de especies con respecto a Mahahual. Mientras en el área continué el desarrollo costero y la pesca en el área, es necesario un análisis más extensivo (escala temporal y espacial) del funcionamiento de ambas lagunas arrecifales.

INTRODUCTION

Seagrass beds are among the most productive aquatic ecosystems in the world (Duarte and Chiscano 1999) and support diverse communities of fishes and invertebrates. These habitats are an important component of the tropical marine environment, and are linked to mangrove and coral reef habitats through fluxes of nutrients and organisms (Parrish 1989, Adams et al. 2006).

Human use or alteration of back-reef biotopes may change their ecological functioning. Coastal development and tourist use of back-reef environments have the potential to degrade habitat through loss of structural complexity or decreased food quality. Globally, seagrass coverages have declined dramatically associated with human environmental degradation (Orth et al. 2006). Also, fishing that often targets larger piscivores may lead to shifts in trophic structure and subsequent community cascades in coastal systems (Chiappone et al. 2000, Graham et al. 2003, Mumby et al. 2006). Understanding the effects of anthropogenic impacts on habitat function becomes a high priority as humans continue to alter many habitats important to ecologically and economically valuable species.

The Mexican Caribbean supports the northern extent of the Mesoamerican Barrier Reef Tract. The Parque Nacional Arrecifes de Xcalak is located on the southern Caribbean coast of Mexico, and development and fishing are restricted within this reserve. However, moderate fishing pressure still exists within the park boundaries as much of the town of Xcalak relies on artesanal fishing. Mahahual is an unprotected location with increasing tourist use after the construction of a cruise ship pier in 2000. Fishing pressure has declined in Mahahual as tourism has taken over as the primary economic activity. This provides the opportunity to compare sites within the reserve with comparatively less coastal development to sites at an unprotected location where coastal development and use of the reef lagoon has dramatically increased. As development continues to threaten coastal ecosystems and

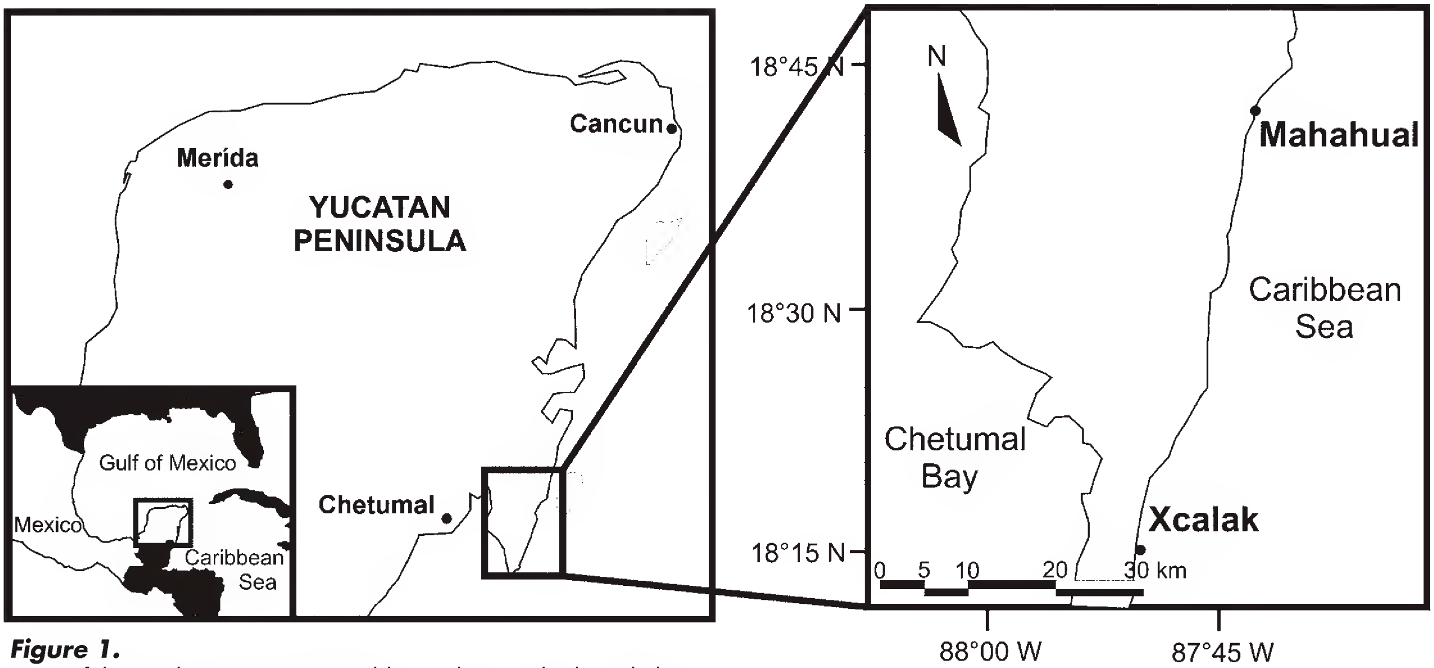

Figure 1.
Map of the southern Mexican Caribbean showing both study locations.

fishing pressure persists in this region, understanding the potential impacts on back-reef habitats becomes imperative.

Few studies have surveyed seagrass fish communities explicitly in the southern Mexican Caribbean. Chitarro et al. (2005) censused seagrass and mangrove habitats at Mahahual and found that juvenile reef fish densities in seagrass were lower than observed juvenile densities in mangroves. Núñez-Lara and Arias-Gonzalez (1998) surveyed lagoon fish communities at Mahahual; however they did not distinguish between lagoon habitat types (i.e. seagrass, sand, patch reefs) in their surveys. Castro-Perez (1998) censused back-reef fish communities at Mahahual, but grouped sand and seagrass habitat together. The authors are not aware of any studies which have surveyed seagrass fish communities in Xcalak. Therefore, overall, little is known about seagrass fish communities in this region.

Most studies of fish communities associated with seagrass habitat have been conducted during the day. However, previous studies of tropical seagrass habitats indicate that nocturnal fish communities may differ substantially from diurnal communities (Weinstein and Heck 1979, Robblee and Zieman 1984, Kopp et al. 2007). Fishes from surrounding habitats, such as coral reefs and mangroves, are known to migrate into seagrass habitat at night to feed (Ogden and Ehrlich 1977, Burke 1995).Therefore, in order to accurately assess the value of seagrass habitat, it is imperative to consider both diurnal and nocturnal communities of fishes that may associate with this critical habitat.

The primary objective of this study is to make a preliminary comparison of seagrass fish community structure, species richness and fish density between two back-reef lagoons with different levels of protection and human use. Additionally, a second objective is to investigate day-night shifts in seagrass fish communities at these sites.

Methods

Study Area

Both study locations are on the southern Caribbean coast of Quintana Roo, Mexico (Figure 1). Xcalak (18°15' N, 87°50' W) is located within Parque Nacional Arrecifes de Xcalak, a marine protected area managed by the Comisión Nacional de Áreas Naturales Protegidas (CONANP). The reserve encompasses 179.49 km^2 of terrestrial and marine habitats. Little coastal development exists within the park and tourist use is low. Fishing is allowed with permits within the reserve, and un-permitted fishing with nets is common. The study location was 2 km south of the town of Xcalak. Mahahual (18°42' N, 87°42' W) is located 50 km north of Xcalak and is unprotected by conservation management regulations. A cruise ship pier was built in Mahahual in 2000, which resulted in increased tourist traffic and development. Sites surveyed at Mahahual were located about 2 km south of most of the town's development.

All sites at both locations were in continuous seagrass habitat (dominated by *Thalassia testudinum*) with *Syringodium filiforme* and macroalgae (including *Laurencia* sp., *Halimeda* sp. *Penicillis capitatus*, *Dictyota* sp., *Padina* sp. *Amphiroa* sp. and *Caulerpa* sp.), and were characterized by sandy bottom. At Xcalak sites were 1.0 to 1.6 m in depth and located within the back-reef lagoon (about 1 km wide) adjacent to fringing mangroves. At Mahahual, all sites were located in seagrass habitat 1.0 to 2.0 m deep adjacent to sandy shore in a reef lagoon (lagoon width ranged from 0.25 to 0.45 km). Seagrass beds at Mahahual were patchier than those

surveyed at Xcalak (L.A. Yeager, personal observation).

Visual Surveys

To determine the composition of the seagrass fish community, visual censuses were completed during the daytime and nighttime in November and December 2006. At Xcalak and Mahahual, 15 and 19 sites were surveyed during the day and 9 and 5 sites were surveyed at night, respectively. All diurnal censuses were completed between 1120 and 1520 h and nocturnal surveys between 1830 and 2020 h. A dive light was used to illuminate the transect during nighttime censuses. Visual surveys were conducted along 20 m belt transects of 2-m width (modified from Brock 1954). All surveys were performed by the first author while snorkeling as follows: the transect line was started from a haphazardly selected point in continuous seagrass habitat and laid out perpendicular to shore. All transect starting points were at least 20 m apart and no transects overlapped within time of day (some transects between day and night in the same locations may have overlapped). Fish species abundance and estimated total length (TL) in 5-cm size classes (e.g., 0-4.9 cm, 5-9.9 cm, 10-14.9 cm) were recorded on dive slates with 5-cm increments marked on the side to aid in estimation of fish size. Members of the species *Sparisoma radians*, *S. aurofrentatum*, and *Nicholsina usta* were grouped into a Scaridae complex due to difficulties distinguishing between juveniles.

Data Analysis

For comparisons of overall community structure, diurnal fish community data were square-root transformed to increase contribution of less abundant species (Clarke and Green 1988). The Bray-Curtis index was used to create a similarity matrix of species-specific abundance data (Clarke 1993). A multi-dimensional scaling (MDS) plot

TABLE 1. *List of observed taxa with mean density (# of individuals/40m² ± standard error) and percent community composition by location and time of day. n = number of transects per time of day and location.*

		XCALAK				MAHAHAUL			
		Day (n = 15)		Night (n = 9)		Day (n = 19)		Night (n = 5)	
Taxa	Common Name	$\bar{x}$ Density	Percent	$\bar{x}$ Density	Percent	$\bar{x}$ Density	Percent	$\bar{x}$ Density	Percent
Muraenidae									
Gymnothorax vicinus	Purplemouth moray	0	0	0	0	0	0	0.20 ± 0.20	20.0
Ophichthidae									
Myrichthys breviceps	Sharptail eel	0	0	0	0	0.11 ± 0.07	1.2	0	0
Synodontidae									
Synodus intermedius	Sand diver	0.07 ± 0.07	0.4	0	0	0	0	0	0
Carangidae									
Carangoides ruber	Bar jack	0.33 ± 0.33	2.2	0	0	0.74 ± 0.49	8.3	0	0
Caranx crysos	Blue runner	0	0	0	0	0.11 ± 0.11	1.2	0	0
Lutjanidae									
Lutjanus griseus	Gray snapper	0.20 ± 0.14	1.3	0	0	0	0	0	0
Lutjanus synagris	Lane snapper	0.40 ± 0.13	2.6	0	0	0.26 ± 0.13	3.0	0	0
Ocyurus chrysurus	Yellowtail snapper	1.60 ± 0.85	10.3	0	0	0.63 ± 0.22	7.1	0	0
Gerridae									
Eucinostomus sp.	Mojarra	0	0	0	0	0.16 ± 0.16	1.8	0	0
Gerres cinereus	Yellowfin mojarra	0	0	0	0	0.32 ± 0.22	3.6	0	0
Haemulidae									
Haemulon plumieri	White grunt	0.13 ± 0.13	0.9	0.22 ± 0.15	20.0	0.05 ± 0.05	0.6	0	0
Haemulon scirus	Blue-stripped grunt	0	0	0	0	1.47 ± 1.26	16.6	0.20 ± 0.20	20.0
Mullidae									
Pseudupeneus maculatus	Yellowtail goatfish	0.8 ± 0.3	2.2	0	0	0.16 ± 0.12	1.8	0	0
Chaetodontidae									
Chaetodon capristratus	Foureye butterflyfish	0	0	0	0	0	0	0.20 ± 0.20	20.0
Chaetodon ocellatus	Spotfin butterflyfish	0	0	0	0	0.05 ± 0.05	0.6	0	0
Pomecentridae									
Abudefduf saxatilis	Sergeant major	0	0	0	0	0.05 ± 0.05	0.6	0	0
Labridae									
Halichoeres bivittatus	Slippery dick	9.60 ± 1.26	62.1	0	0	1.42 ± 0.45	16.0	0	0
Halichoeres poeyi	Blackear wrasse	0.07 ± 0.07	0.4	0	0	0.79 ± 0.28	8.9	0	0
Scaridae									
Scaridae complex	Parrotfish	2.40 ± 0.63	15.5	0	0	2.21 ± 0.60	24.9	0.20 ± 0.20	20.0
Scarus iserti	Striped parrotfish	0.07 ± 0.07	0.4	0	0	0	0	0	0
Scarus taeniopterus	Princess parrotfish	0.27 ± 0.18	1.7	0	0	0	0	0	0
Sparisoma rubripinne	Redfin parrotfish	0	0	0	0	0.26 ± 0.17	3.0	0	0
Acanthuridae									
Acanthurus coeruleus	Blue tang	0	0	0	0	0.05 ± 0.05	0.6	0	0
Sphyraenidae									
Sphryaena barracuda	Great Barracuda	0	0	0.11 ± 0.11	10.0	0	0	0	0
Tetradontidae									
Sphoeroides spengleri	Bandtail puffer	0	0	0	0	0.05 ± 0.05	0.6	0	0
Diodontidae									
Diodon holocanthus	Long-spine porcupinefish	0	0	0.78 ± 0.32	70.0	0	0	0.20 ± 0.20	20.0
Total		**15.47 ± 2.13**		**1.11 ± 0.42**		**8.89 ± 1.70**		**1.00 ± 0.50**	

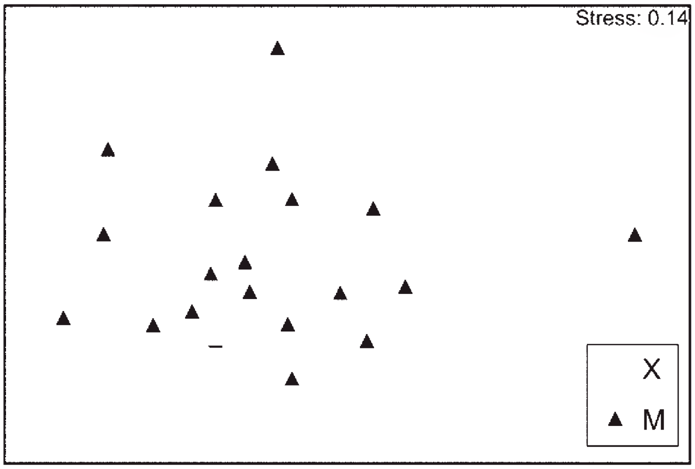

Figure 2.
MDS plot comparing community structure between study locations based on Bray-Curtis similarity matrix of diurnal fish community data (per species density in fish/40m²). Two overlapping points in the center of the cluster of gray triangles both represent transects at Xcalak. M=Mahahual and X=Xcalak.

was employed based on the similarity matrix to graphically explore differences in seagrass fish communities. An Analysis of Similarity (ANOSIM) was performed to test for statistical differences in fish communities between locations. Importance of individual taxa in contributing to differences between locations was determined with a Similarity-Percentages (SIMPER) analysis of square-root transformed abundance data (Clarke 1993, Primer© 5).

Non parametric statistics were used for all comparisons of fish abundance, species richness and length frequency because variables were not normally distributed despite numerous transformations (Kolmogorov-Smirnov normality test, $p < 0.05$ in all cases). Mean total fish abundance and species richness per transect were each compared between locations for diurnal communities with a Kruskal-Wallis ANOVA. For diel comparisons, data were grouped between locations by time of day. We feel this grouping of data was justified because of the lower number of nocturnal surveys and great differences between nocturnal and diurnal communities. Total fish abundance and species richness per transect between night and day were compared with a Kruskal-Wallis ANOVA.

Results

A total of 417 individuals representing 28 taxa from 16 families were observed in seagrass habitats at the two study locations during diurnal and nocturnal censuses (Table 1). The most abundant families included parrotfishes (Scaridae), wrasses (Labridae), snappers (Lutjanidae), grunts (Haemulidae), and jacks (Carangidae).

Daytime fish communities differed among locations (ANOSIM, Global $R = 0.21$, $p = 0.002$, Figure 2). Sites within Xcalak had greater similarity (57%) than sites within Mahahual (25%). Sites at Xcalak were characterized by two primary taxa (*Halichoeres bivitattus* and Scaridae complex, 77.6% of all fishes observed) while sites at Mahahual were dominated by Scaridae complex, *H. bivittatus*, *H. poeyi*, *Ocyurus chrysurus*, and *Carangoides ruber*, with these taxa making up 90% of all fishes observed (SIMPER). Sites at Xcalak were differentiated from those at Mahahual by the increased relative importance of *H. bivitattus*, Scaridae complex, *O. chrysurus* and *Lutjanus synagris* and decreased relative importance of *H. poeyi* and *C. ruber* (SIMPER). Even though *O. chrysurus* was not one of the two most abundant species making up 90% of the fish community at Xcalak, it still was more abundant at this location than at Mahahual (Table 1). Daytime fish abundance at Xcalak ($\bar{x} = 15.5 \pm 2.1$ fish/40m²) was greater than that at Mahahual ($\bar{x} = 8.9 \pm 1.7$ fish/40m², Kruskal-Wallis, $H = 6.806$, $p = 0.009$, Table 1). However, species richness did not differ between regions in the daytime ($\bar{x}_{xcalak} = 3.2 \pm 0.4$ fish species/transect, $\bar{x}_{mahahual} = 3.7 \pm 0.5$ fish species/transect, Kruskal-Wallis, $H = 0.150$, $p = 0.698$).

Families observed during nocturnal censuses included porcupinefishes (Diodontidae), grunts, parrotfishes, barracudas (Sphyraenidae), butterflyfishes (Chaetodontidae), and moray eels (Muraenidae). Nighttime fish communities exhibited much lower abundance (for both locations combined: $\bar{x} = 1.1 \pm 0.3$ fish/40m², Kruskal-Wallis, $H = 26.625$, $p < 0.001$) and lower species richness ($\bar{x} = 0.9 \pm 0.3$ fish species/transect, Kruskal-Wallis, $H = 19.798$, $p < 0.001$) when compared to daytime communities in this study.

Discussion

Diurnal communities of seagrass fishes differed between two back-reef lagoons with physical differences that included different levels of protection. The anthropogenic influences at both sites may partially explain the observed differences in fish communities. Greater environmental degradation, reflected in patchy seagrass habitat, at Mahahual may have led to lower abundance of fishes within this back-reef lagoon. Mahahual village has been recently urbanized for receiving thousands of tourists brought in by cruise ships. The beach was increased with dredged sand from the reef lagoon, the reef lagoon channel was deepened for boat and personal water craft transit, seagrass beds were removed, and the seascape was transformed with construction of a pier, small restaurants, shops and cabins. Following construction of the cruise ship pier, coral cover has decreased and algal cover has increased on coral reefs in Mahahual (Arias-Gonzalez et al., unpublished data), a sign of habitat degradation. However, no historical data related to seagrass fish communities or seagrass coverage at Mahahual are available for comparison to assess possible declines in abundance or shifts in community structure.

The greater fish density at Xcalak was mainly attributed to the greater abundance of *H. bivittatus*. Although this species is typically considered to be a habitat generalist (Grat-

wicke et al. 2006), the more extensive, less-disturbed seagrass habitat at Xcalak may have been preferable to this species.

Differences in contiguous habitats at the study locations may also account for differences in fish communities. Proximity to surrounding habitats affects the distributions of various fish species in back-reef environments (Drew 2006). Previous studies of seagrass fishes have found that proximity to mangroves and/or coral reef habitats may affect community structure or fish abundances (Robblee and Zieman 1984, Baelde 1990, Kopp et al. 2007). Mumby et al. (2004) found that coral reef sites adjacent to mangroves supported much greater biomass of fishes compared to those without mangroves nearby. Similarly, Kopp et al. (2007) observed greater fish density and biomass in seagrass habitat located adjacent to mangroves when compared to seagrass habitat near a coral reef. However, Baelde (1990) reported greater catch and greater species richness in a seagrass beds located in close proximity to mangrove and coral reef habitat compared to seagrass habitat only associated with mangroves. Seagrass beds at Xcalak are bordered by mangroves which may have contributed to the higher abundance of fishes found at this site. Also, the presence of *L. griseus* at Xcalak is likely due to the presence of mangroves at this location as this species is known for its association with mangrove habitats (e.g., Nagelkerken et al. 2000a, Gratwicke et al. 2006).

In contrast, the close proximity of reef habitat to Mahahual seagrass beds may have influenced community composition towards reef-associated species. The lagoon is narrower at Mahahual than at Xcalak, which may lead to increased connectivity between the reef and seagrass habitat at Mahahual. For example, *C. ruber*, a species that travels between reef and lagoon habitats, was more abundant and a more important component of the fish community at Mahahual.

Even though fish densities are often reported to be lower in seagrass habitat than in surrounding coral dominated areas, total habitat area must be taken into account (Nagelkerken et al. 2000b, Mateo and Tobias 2004). Seagrass habitat is often quite extensive in back-reef lagoons when compared to the coverage of other habitat types (e.g., patch reefs). Therefore, even though fish densities are lower, the total contribution of seagrass beds as habitat may be greater (Nagelkerken et al. 2000b). Also, juveniles of ecologically and commercially important species (e.g., *O. chrysurus*, *L. griseus*, *L. synagris*) were observed in seagrass habitat, suggesting this habitat may serve as a nursery area for these species. Lesser abundance of predators may make seagrass meadows the preferred feeding/sheltering habitat for a number of fishes (Shulman 1985). Additionally, habitat use only provides one measure of the value of a habitat. The function of seagrass habitat in Mahahual and Xcalak in terms of providing refuge from predation, food sources, or connectivity to other habitats is unknown. More extensive surveys of daytime and nighttime fish communities, as well as investigation of other aspects of ecosystem functions of these habitats, are needed to fully understand the importance of seagrass habitat in these back-reef systems.

Diurnal seagrass fish communities of the southern Mexican Caribbean observed in this study were similar to assemblages in other regions of the Caribbean, being dominated by wrasses, parrotfishes, snappers and grunts (Weinstein and Heck 1979, Nagelkerken et al. 2000b, Mateo and Tobias 2004). However, most studies only survey daytime fish communities due to logistical difficulties associated with nocturnal sampling. The preliminary surveys in this study suggest that seagrass habitat may not be as important during the nighttime in terms of total fish density or species richness. Kopp et al. (2007) also observed low abundance of nocturnal fishes in seagrass habitat near mangroves when compared to seagrass near coral reefs or to diurnal abundance. Likewise, Nagelkerken et al. (2000c) found lower fish density and species richness during nighttime in seagrass habitat in Spanish Water Bay, Curaçao than during the day, but suggested that seagrass was an important nighttime feeding habitat for snappers and grunts. Weinstein and Heck (1979) reported increased abundance of adult grunts and snappers in seagrass habitat at night and found similar or greater abundance of fishes at night than during the daytime. In this study, grunts were observed at night and snappers were observed outside of the transects at night. However, members of both of these families are highly mobile, and true patterns of their habitat use may not have been detected with the lower number of nocturnal surveys in this study. Members of other diurnally dominant fauna (e.g., wrasses) were not observed during the nighttime in this study. Similarly, Robblee and Zieman (1984) found that diurnal fish communities in seagrass habitat in Tague Bay, St. Croix were dominated by small permanent residents of the seagrass bed, whereas nocturnal fish communities were dominated by predatory reef species. There appears to be a shift in fish communities between night and day, emphasizing the importance of considering diel changes in habitat use to gain a more complete understanding of the functioning of seagrass beds at these two locations.

Underwater visual census is a widely used technique for surveying shallow-water fish communities. The authors acknowledge that some biases associated with this technique do exist (e.g., observer effect, Samways and Hatton 2001), but efforts to minimize effects of the observer and transect line were made. However, abundance of more cryptic species that hide within the seagrass canopy may be underestimated. In this study system the relatively clear water and short seagrass canopy should have reduced this potential bias. Also, the relative efficacy of underwater visual census in quantifying fish abundance in seagrass habitat between day and night is not known. Some species may avoid the dive lights necessary for nocturnal surveys. However, a previous

study using visual census to quantify diurnal and nocturnal fish communities in a variety of back-reef habitats including seagrass suggested dive lights did not seem to modify the behavior of most nocturnal species (Nagelkerken et al. 2000c).

Detailed conclusions about nocturnal community structure at these sites are difficult to reach when considering the confounding factors of lower numbers of nocturnal surveys and decreased nocturnal fish abundance. While these potential biases must be considered, qualitative differences in fish community composition at a family level as well as differences in fish abundance and species richness between night and day were so great that the overall patterns are believed to be real.

A limitation of this study is the fact that both anthropogenic influences and habitat characteristics varied between locations, so it is difficult to attribute differences in fish communities to any one factor. While the results presented in this paper are preliminary, and based on data collected during only one season, they do suggest both anthropogenic pressures and habitat differences between locations are affecting the fish communities. Additionally, this study provides an initial survey of seagrass fish communities in a little studied area that will likely continue to undergo change with increasing anthropogenic pressures. As coastal development and tourism continue to increase at both locations, monitoring of fish communities and bentho is recommended to better evaluate potential threats and changes in seagrass ecosystems.

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Literature Cited

Adams, A.J., C.P. Dahlgren, G.T. Kellison, M.S. Kendall, C.A. Layman, J.A. Ley, I. Nagelkerken, and J.E. Serafy. 2006. Nursery function of tropical back-reef systems. Marine Ecology Progress Series 318:287-301.

Baelde, P. 1990. Differences in the structures of fish assemblages in *Thalassia testudinum* beds in the Guadeloupe, French West Indies, and the ecological significance. Marine Biology 105:163-173.

Brock, R.E. 1954. A preliminary report on a method of estimating reef fish populations. Journal of Wildlife Management 18:297-308.

Burke, M.L. 1995. Nocturnal foraging habitats of French and bluestriped grunts, *Haemulon flavolineatum* and *H. sciurus*, at Tobacco Caye, Belize. Environmental Biology of Fishes 42:365-374.

Castro-Perez, J.M. 1998. Estructura de la comunidad de peces asociada al arrecife de Mahahual, Quintana Roo, Mexico. Master's Thesis. Centro de Investigación y de Estudios Avanzados del I.P.N, Merida, Yucatán, Mexico. 66 p.

Chiappone, M., R. Sluka, and K. Sullivan-Sealey. 2000. Groupers (Pisces: Serranidae) in fished and protected areas of the Florida Keys, Bahamas and northern Caribbean. Marine Ecology Progress Series 198:261-272.

Chittaro, P.M., P. Usseglio, and P.F. Sale. 2005. Variation in fish density, assemblage composition and relative rates of predation among mangrove, seagrass and coral reef habitats. Environmental Biology of Fishes 72:175-187.

Clarke, K.R. 1993. Non-parametric multivariate analyses of changes in community structure. Australian Journal of Ecology 18:117-143.

Clarke, K.R. and R.H. Green. 1988. Statistical design and analysis for a 'biological effects' study. Marine Ecology Progress Series 46:213-226.

Drew, C.A. 2006. Spatial ecology of reef fish in backreef and coral reef habitats. Ph.D. Thesis. North Carolina State University, Raleigh, NC, USA, 124 p.

Duarte, C.M. and C.L. Chiscano. 1999. Seagrass biomass and production: a reassessment. Aquatic Botany 65:159-174.

Graham, N.A.J., R.D. Evans, and G.R. Russ. 2003. The effects of marine reserve protection on the trophic relationships of reef fishes on the Great Barrier Reef. Environmental Conservation 30:200-208.

Gratwicke, B., C. Petrovic, and M.R. Speight. 2006. Fish distribution and ontogenetic habitat preferences in non-estuarine lagoons and adjacent reefs. Environmental Biology of Fishes 76:191-210.

Kopp, D., Y. Bouchon-Navaro, M. Louis, and C. Bouchon. 2007. Diel differences in the seagrass fish assemblages of

a Caribbean island in relation to adjacent habitat types. Aquatic Botany 87:31-37.

Mateo, I. and W.J. Tobias. 2004. Survey of nearshore fish communities on tropical backreef lagoons on the southeastern coast of St. Croix. Caribbean Journal of Science 40:327-342.

Mumby, P.J., C.P. Dahlgren, A.R. Harborne, C.V. Kappel, F. Micheli, D.R. Brumbaugh, K.E. Holmes, J.M. Mendes, K. Broad, J.N. Sanchirico, K. Buch, S. Box, R.W. Stoffle, and A.B. Gill. 2006. Fishing, trophic cascades, and the process of grazing on coral reefs. Science 311:98-101.

Mumby, P.J., A.J. Edwards, J.E. Arias-Gonzalez, K.C. Lindeman, P.G. Blackwell, A. Gall, M.I. Gorczynska, A.R. Harborne, C.L. Pescod, H. Renken, C.C.C. Wabnitz, and G. Llewellyn. 2004. Mangroves enhance the biomass of coral reef fish communities in the Caribbean. Nature 427:533-536.

Nagelkerken, I., G. Van der Velde, M.W. Gorissen, G.J. Meijer, T. van't Hof, and C. den Hartog. 2000a. Importance of mangroves, seagrass beds and the shallow coral reef as a nursery for important coral reef fishes, using a visual census technique. Estuarine, Coastal and Shelf Science 51:31-44.

Nagelkerken, I., M. Dorenbosch, W.C.E.P. Verberk, E. Cocheret de la Moriniere, and G. van der Velde. 2000b. Importance of shallow-water biotopes of a Caribbean bay for juvenile coral reef fishes: patterns in biotope association, community structure and spatial distribution. Marine Ecology Progress Series 202:175-192.

Nagelkerken, I., M. Dorenbosch, W.C.E.P. Verberk, E. Cocheret de la Moriniere, and G. van der Velde. 2000c. Day-night shifts of fishes between shallow-water biotopes of a Caribbean bay, with emphasis on the nocturnal feeding of Haemulidae and Lutjanidae. Marine Ecology Progress Series 194:55-64.

Núñez-Lara, E. and J.E. Arias-González. 1998. The relationship between reef fish community structure and environmental variables in the southern Mexican Caribbean. Journal of Fish Biology 53:209-221.

Ogden, J.C. and P.R. Ehrlich. 1977. The behavior of heterotypic resting schools of juvenile grunts (Pomadasyidae). Marine Biology 42:273-280.

Orth, R.J., T.J.B. Carruthers, W.C. Dennison, C.M. Duarte, J.W. Fourqurean, K.L. Heck Jr., A.R. Hughes, G.A. Kendrick, W.J. Kenworthy, S. Olyarnik, F.T. Short, M. Waycott, and S.L. Williams. 2006. A global crisis for seagrass ecosystems. Bioscience 56:987-996.

Parrish, J.D. 1989. Fish communities of interacting shallow-water habitats in tropical oceanic regions. Marine Ecology Progress Series 58:143-160.

Robblee, M.B. and J.C. Zieman. 1984. Diel variation in the fish fauna of a tropical seagrass feeding ground. Bulletin of Marine Science 34:335-345.

Samways, M.J. and M.J. Hatton. 2001. An appraisal of two coral reef rapid monitoring manuals for gathering baseline data. Bulletin of Marine Science 69:471-485.

Shulman, M.J. 1985. Recruitment of coral reef fishes: effects of distribution of predators and shelter. Ecology 66:1056-1066.

Weinstein, M.P. and K.L. Heck Jr. 1979. Ichthyofauna of seagrass meadows along the Caribbean coast of Panama and in the Gulf of Mexico: Composition, structure and community ecology. Marine Biology 50:97-107.

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SHELL UTILIZATION PATTERN BY THE HERMIT CRAB *ISOCHELES SAWAYAI* FOREST AND SAINT LAURENT, 1968 (ANOMURA, DIOGENIDAE) FROM MARGARITA ISLAND, CARIBBEAN SEA, VENEZUELA

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Abstract: *Isocheles sawayai* is a hermit crab that is occasionally mentioned in the literature, and recently its distribution was extended to Venezuelan waters. Because no information on the biology and shell use patterns of this species inhabiting Caribbean waters is available, we provide the first information on shell occupation patterns of *I. sawayai* from Venezuela. Specimens were collected monthly from January to December 2000 along the sandy shore of Margarita Island, Venezuela. The 942 specimens collected showed different shell use patterns between the sexes and according to the reproductive condition of the females. The gastropods *Leucozonia nassa* (37.37%), *Engoniophos unicinctus* (25.37%), *Nassarius vibex* (4.88%), *Melongena melongena* (4.25%), and *Stramonita haemastoma* (3.82%) represent 76% of the total occupied shells. Of the total of 26 different shell species occupied by *I. sawayai*, males were found occupying 21, while females were found occupying all 26 shell species. In general, both sexes most frequently occupied *L. nassa* and *E. unicinctus*. However, the percentage of females occupying these shells was significantly higher than that of the males. Regression analyses showed the best correlation between crab size, shell aperture width, and shell internal volume. The current comparative investigation, in combination with other South Atlantic populations of *I. sawayai*, provided further evidence of shell use adaptation in hermit crabs from different areas, and increases our insight into shell use of shallow-water hermit crabs.

Introduction

Hermit crabs are an interesting group from a biological and evolutionary viewpoint, especially in regard to their intriguing mechanisms of shell use. The borrowed shells of gastropods provide protection against predators and physical stress, and often constitute a limiting resource for hermit crabs (e.g., Reese 1969, Vance 1972, Fotheringham 1976a, Bertness 1981a) in terms of growth, reproduction, and social behavior.

Although many studies have been published on hermit crab shell occupation worldwide, the parameters by which a particular shell is chosen by a hermit crab are not completely known (Meireles and Mantelatto 2005). Studies on gastropod shell availability, patterns of shell use and selection, and the relationship between these factors constitute an initial part of a long-term effort undertaken to identify and clarify important parameters affecting this process (Mantelatto and Meireles 2004). The patterns of shell use vary among hermit crab populations and are influenced by several factors, such as the type and size of available shells, the inhabited area (intertidal or sublittoral area), and the hermit crabs' shell preference (Garcia and Mantelatto 2000, Mantelatto and Garcia 2000).

The genus *Isocheles* Stimpson, 1858 is known taxonomically, but there is a lack of basic biological information. For the five species in the genus reported in American tropical and subtropical waters, there are only brief references concerning their distributions (see Forest and Saint Laurent 1968, Nucci and Melo 2000, Guzmán 2004), and a recent study on molecular phylogeny (Mantelatto et al. 2006). In relation to *Isocheles sawayai* Forest and Saint Laurent 1968, the available information is restricted to data on specimens from the Brazilian coast that deals with the morphology of larval stages (Negreiros-Fransozo and Hebling 1983), shell use (Pinheiro et al. 1993, Fantucci et al. 2008), and records of intersex individuals (Fantucci et al. 2007).

Although shell use by hermit crabs has been examined in other areas of the world (see Mantelatto and Garcia 2000 for review), to our knowledge no detailed and systematic study has been carried out on the hermit crab fauna of the Caribbean or Atlantic region of Venezuela. Here we report the first observations on the patterns of gastropod shell occupation by a population of *I. sawayai* inhabiting the sandy shore of Isla Margarita, Venezuela, with emphasis on morphometric relationships between hermit crabs and their shells.

Materials and Methods

Study Area

La Restinga Beach is located in a northern cornice of Margarita Island, Venezuela on the Caribbean Sea (10°57' N - 11°03' S, 64°01' - 64°12' W; Figure 1) and is open to northwest trade winds. The beach is formed by a flat sand fringe that separates a hypersaline lagoon from the ocean, has a total surface area of about 30 km^2, and is the largest beach in Nueva Esparta Province. The mean water temperature during collections was 26 °C and the salinity was 34 ppt.

Sampling Procedures

Specimens of *I. sawayai* were collected monthly during daytime on La Restinga Beach from January to December 2000 at random locations along the beach over a distance of about 200 m. A minimum of 70 individual hermit crabs were captured by hand each month. After collection, the animals were transported to the Crustacean Laboratory of the Universidad de Oriente and preserved in a 5% solution of formalin in sea water. Each individual was removed from its occupied shell, sexed, wet weighed (WW, g), and the cephalothoracic shield length (CSL, mm) was measured to the nearest 0.05 mm using a caliper. The reproductive condition of each female (ovigerous, non-ovigerous) was also recorded.

Shell Study

Shell species were identified according to the descriptions in Warmke and Abbott (1962), Abbott (1974), and Morris (1975), and confirmed by a specialist. The measurements were made according to Imafuku and Ando (1999) and Mantelatto and Garcia (1999): Total Shell Length (TSL, mm); Shell Maximum Width (SMW, mm); and Shell Dry Weight (SDW, g). Shell Internal Volume (SIV, cc^3) was measured by the method suggested by Bertness (1981b), and involved determining the volume of sand (known weight) required to fill an empty shell. Shell Angle Tip (SAT, degree) was measured according to Asakura (1995). An empirical scale was used to evaluate the physical state of shell condition, from 1 for perfect shell condition to 6 for severely damaged.

Data Analysis

To determine correlations among hermit crab dimensions and shell variables, regressions $Y = a \bullet X^b$ and correlation coefficients were calculated. The percentage of occupied gastropod shells was estimated based on the total number of individuals collected. The Shannon-Weaver Diversity Index (Margalef 1974), based on the number of gastropod species

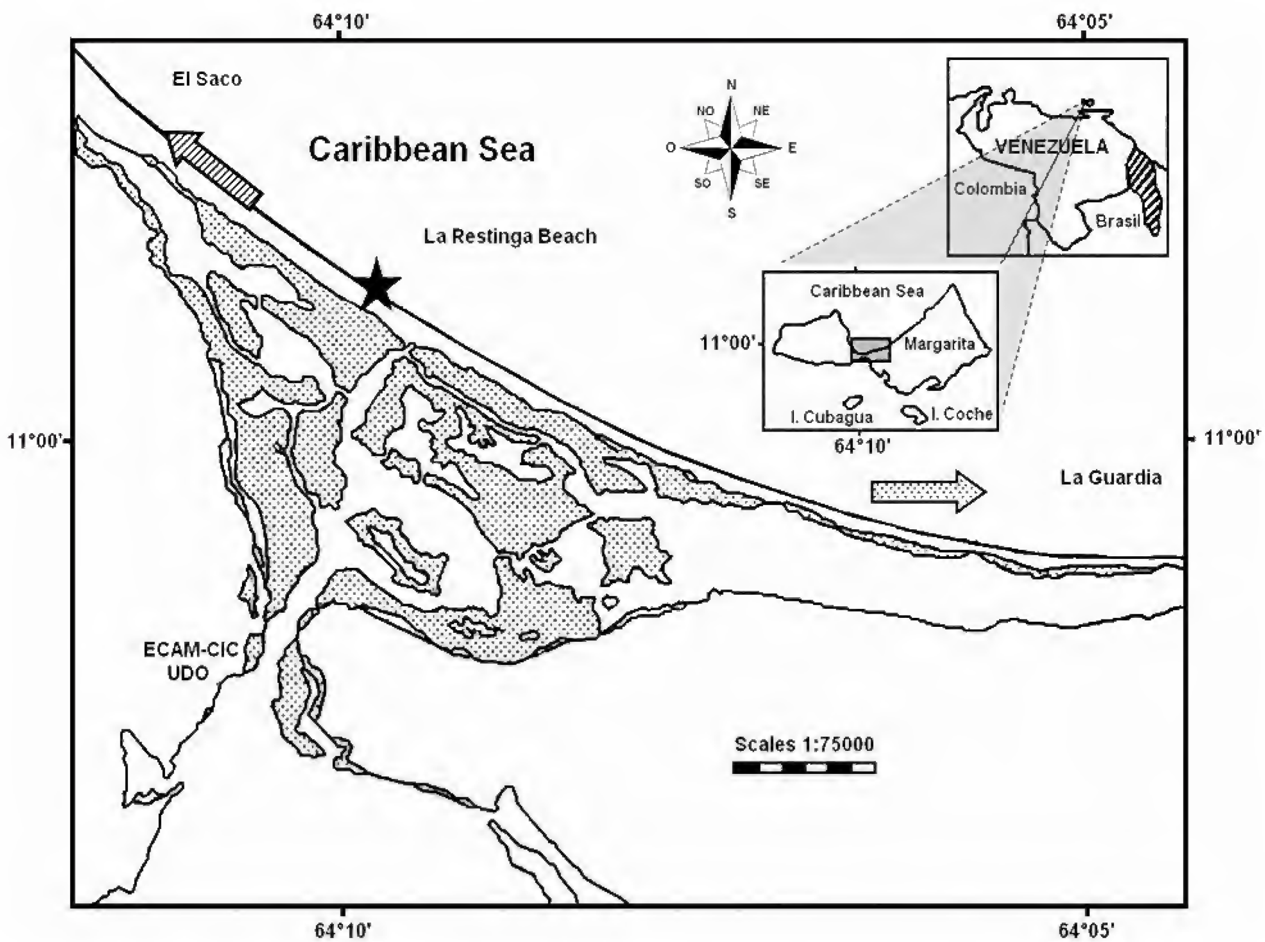

Figure 1.
Map showing the sampling location (star) at an exposed sandy beach at La Restinga, Margarita Island, Venezuela (10°57'N - 11°03'S, 64°01'- 64°12'W). Potential sources of shells: La Guardia (dotted arrow) and El Saco (striped arrow).

TABLE 1. *Species, number, and percentage (monthly and total) of gastropod shells used by Isocheles sawayai from January to December 2000 at La Restinga Beach, Venezuela. Occurrence is expressed as the number of shells found per number of collections.*

Shell Species	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Total**	**Occurrence**	**%**
Leucozonia nassa	52	39	23	14	31	29	26	30	30	42	21	15	352	12/12	37.37
Engoniophos unicinctus	9	13	11	12	15	26	18	22	17	32	38	26	239	12/12	25.37
Nassarius vibex	-	1	4	1	1	5	5	11	2	4	10	2	46	11/12	4.88
Melongena melongena	3	2	5	8	6		5	3	3	2	1	2	40	11/12	4.25
Stramonita haemastoma	10	3	6	5	3	4	1	1	2	1	-	-	36	10/12	3.82
Stramonita rustica	-	6	6	2	-	2	2	1	3	4	2	1	29	10/12	3.08
Natica canrena	2	6	2	3	4	1	4	-	3	1	1	2	29	11/12	3.08
Bursa granularis	4	4	2	2	2	1	2	3	2	2	2	1	27	12/12	2.87
Pisania tincta	5	-	-	2	2	1	-	3	-	5	2	-	20	7/12	2.12
Clathrodrillia gibosa	3	3	3	1	-	-	3	1	-	2	1	1	18	9/12	1.91
Murex chrysostoma	1	3	2	1	1	1	3	-	-	3	-	3	18	9/12	1.91
Turritella variegata	1	-	-	1	1	-	-	3	3	2	3	3	17	8/12	1.80
Conus spp.	-	-	1	-	2	1	2	-	5	-	-	-	11	5/12	1.17
Chicoreus brevifrons	2	-	1	1	1	1	1	-	-	-	1	1	9	9/12	0.96
Marginella prunum	1	-	2	1	1	1	-	1	-	-	1	1	9	8/12	0.96
Others[1]	7	3	1	5	5	3	3	3	2	3	3	4	42	-	4.46
Total number of species per month	**17**	**13**	**14**	**18**	**15**	**15**	**15**	**13**	**12**	**15**	**14**	**14**	**17**		

[1]Species and % occurrence included in Other category: *Fusinus closter* (0.74%), *Cancellaria reticulata* (0.53%), *Olivia reticulatus* (0.42%), *Terebra cinerea* (0.32%), *Fasciolaria tulipa* (0.21%), *Ancilla tankervilli* (0.11%), *Cymatium parthenopeum* (0.11%), *Murex* spp. (0.11%), *Ceritium eburneum* (0.11%), *Conus jaspideus* (0.11%), and *Tegula lividomaculata* (0.11%), 2 unidentified (1.58%).

used by *I. sawayai*, was also calculated. The occupancy of shell species by hermit crabs (males, non-ovigerous females, and ovigerous females) was tested by the chi-square test and the mean size of both sexes was compared by Mann-Whitney U-test (Zar 1996). The level of significance was 0.05 for all tests.

Results

Of 942 hermit crabs collected, 171 were males (18.15%), 164 were non-ovigerous females (17.41%), 600 were ovigerous females (63.69%), and the sex of 7 was undetermined (0.74%). Males and females had a CSL ranging from 1.70 to 7.35 mm and from 2.15 to 8.75 mm, respectively. Males were significantly larger than non-ovigerous ($U = 17167.6$, $p < 0.01$) and ovigerous females ($U = 18985.3$, $p < 0.01$).

Isocheles sawayai occupied shells of 26 gastropod species. *Leucozonia nassa*, *Engoniophos unicinctus*, *Nassarius vibex*, *Melongena melongena*, and *Stramonita haemastoma* represented 75.69% of the total shells obtained. The great majority of hermit crabs (95.55%) occupied 15 of the 26 shell species (Table 1). The shell species used least frequently were *Fusinus closter* (0.74%), *Cancellaria reticulata* (0.53%), *Olivia reticulatus* (0.42%), *Terebra cinerea* (0.32%), *Fasciolaria tulipa* (0.21%), *Ancilla tankervilli* (0.11%), *Cymatium parthenopeum* (0.11%), *Murex* spp. (0.11%), *Ceritium eburneum* (0.11%), *Conus jaspideus* (0.11%), and *Tegula lividomaculata* (0.11%). These 9 species of infrequently used shells plus 2 unidentified ones were designated as "others."

The number of shell species occupied per month by *I. sawayai* ranged from 12 (September) to 18 (April). *Leucozonia nassa*, *S. haemastoma*, and *Stramonita rustica* showed the highest frequency of occurrence (12/12) during the study period (Table 1). Of the total shells analyzed, 16% were covered externally by epibionts (bryozoans in most cases), 50% were generally damaged, and only 34% were in a perfect state of condition.

Morphometric relationships between hermit crabs and used shells were statistically significant, but with low correlation coefficients (Table 2, Figure 2). Of these, the relationships SMW versus CSL and SIV versus CSL turned out to best describe the association between hermit crabs and their shells. Although significant, SAT versus CSL ($r = 0.29$) did not seem to generate much information concerning the population.

Males and females ($\chi^2 = 90.71$; $p < 0.001$; permutation test = 0) as well as ovigerous and non-ovigerous females ($\chi^2 = 30.59$; $p < 0.001$; permutation test = 0) showed different shell use patterns (Figure 3). Males were found occupying 21 of the 26 collected gastropod shells, whereas females (ovigerous

as well as non-ovigerous) used all the shell species collected. The mean Shannon-Weaver diversity (H') of shells occupied by *I. sawayai* was 1.67 ± 0.16 bit/ind and ranged from 1.51 to 2.07 bit/ind. High values of H' indicate the population occupied a wider variety of shell species. In general, males and females of *I. sawayai* inhabit *L. nassa* (19% □; 42% □, respectively) and *E. unicinctus* (16% ♂; 28% □, respectively) most frequently; the percentage of females using these two shells was significantly higher than that of males ($\chi^2 = 25.23$; $p < 0.001$; permutation = 0.006 and $\chi^2 = 14.92$; $p < 0.01$; permutation = 0.05; respectively). The use of *E. unicinctus* by ovigerous and non-ovigerous females was statistically different ($\chi^2 = 16.49$; $p < 0.01$; permutation = 0.01). However, there was no evidence of a difference between the reproductive state of females and their occupation of *L. nassa* ($\chi^2 = 7.09$; $p < 0.31$; permutation = 0.34). The variability of shell species used decreased as the hermit crab size increased (Figure 4). However, in the smallest size class of hermit crab the shell variability was limited by the availability of small shells.

Discussion

In general, *I. sawayai* occupied a wider variety of shell species compared with other hermit crabs from tropical and subtropical areas (Table 3). Although shell availability was not evaluated, intense occupancy of some species of gastropod shells would indicate active selection behavior in *I. sawayai* in the field. *Isocheles sawayai* is a medium-sized hermit crab; this size may increase the possibility of finding adequate shells compared to larger crabs that are forced to look for larger shells.

Judging from the large number of shell species used, it could be expected that high gastropod shell diversity is available at La Restinga Beach; however, neither living gastropods nor empty shells were found in the field during the present study. The present study revealed a high variation of shell use in this hermit crab population. Generally, when there is good availability of resources (*i.e.*, empty shells, live

TABLE 2. *Shell dimensions of gastropods occupied by Isocheles sawayai from January to December 2000 at La Restinga Beach, Venezuela. TSL = Total Shell Length; SMW = Shell Maximum Width; SDW = Shell Dry Weight; SIV = Shell Internal Volume; SAT = Shell Angle Tip; N = Number of Shells; Min = Minimum Value; Max = Maximum Value;* $\bar{X}$ *= Mean; sd = Standard Deviation; CV = Coefficient of Variation.*

	N	Min	Max	$\bar{X}$	sd	CV
TSL (cm)	941	0.005	0.729	0.235	0.068	28.80
SMW (cm)	938	0.019	0.441	0.143	0.043	29.68
SDW (g)	942	0.07	16.90	1.72	1.46	85.01
SIV (cm³)	899	0.02	8.12	0.69	0.86	124.48
SAT (°)	937	12	135	60.60	18.70	30.90

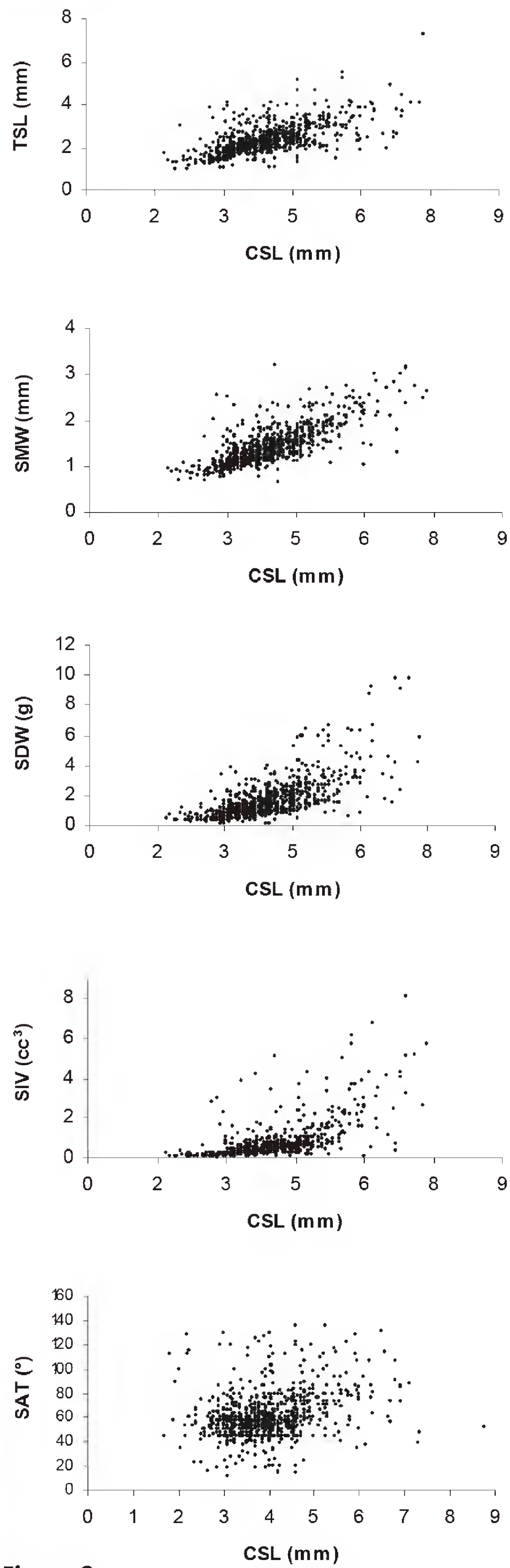

Figure 2.
Regression plots between cephalothoracic shield length (SL) of Isocheles sawayai and total shell length (TSL), shell maximum width (SMW), shell dry weight (SDW), shell internal volume (SIV), and shell angle tip (SAT).

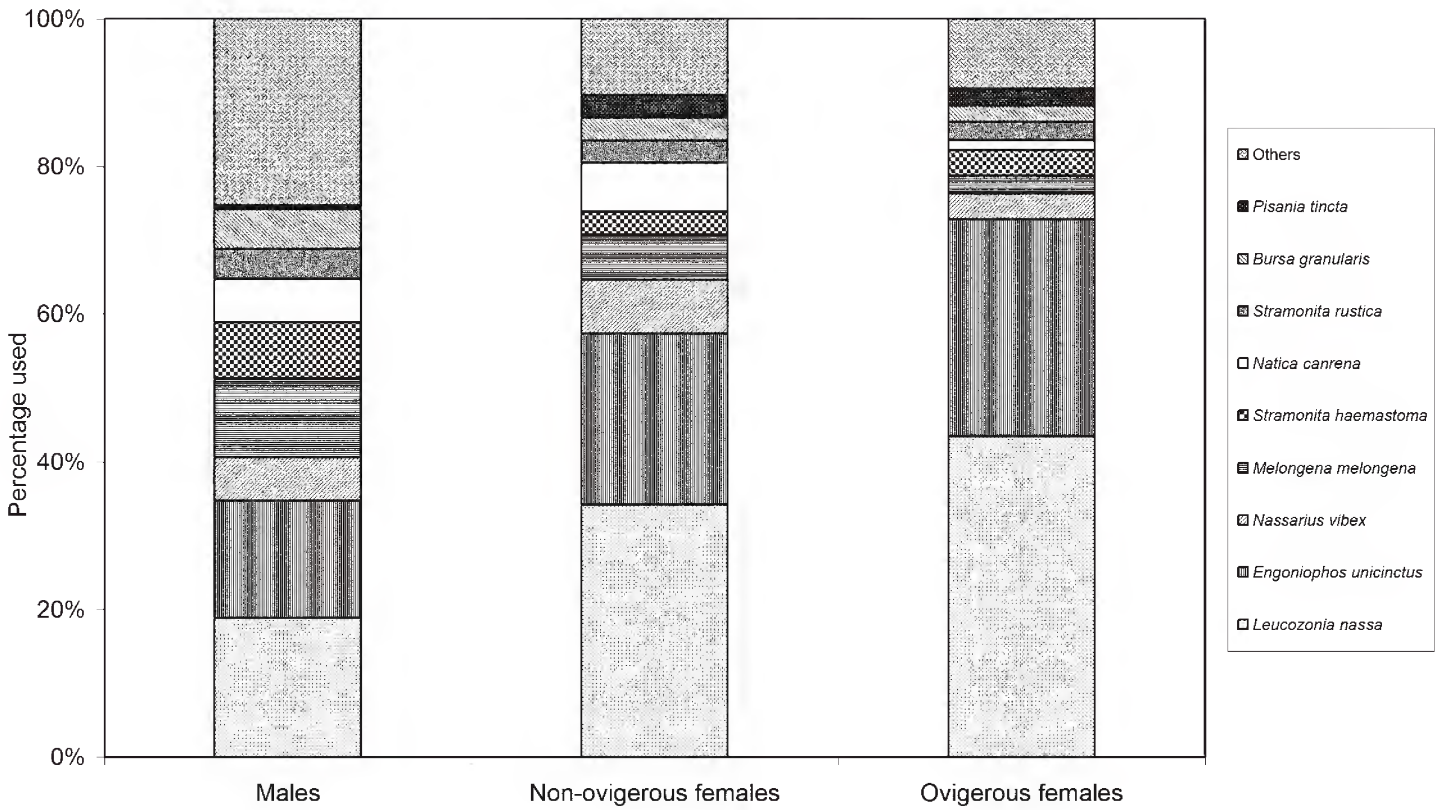

Figure 3.
Percent use by Isocheles sawayai of different species of gastropod shells by demographic category.

gastropods, and high diversity of occupied shells), the most common pattern observed is an occupation of adequate and undamaged shells by the hermit crabs (Mantelatto and Meireles 2004). However, this condition was not observed in the population studied: a large number (76%) of damaged shells were found, and no preference for occupation of these shells by either sex or ovigerous females was observed. A similar pattern has been described for other hermit crabs from different areas worldwide (Vance 1972, Bach et al. 1976, Martinelli and Mantelatto 1999, Mantelatto and Meireles 2004). According to Scully (1979), the occupation of damaged shells increases the probability of interactions and shell exchanges within a population, affecting both protection and reproduction of the resident hermit crabs. For example, in situations where shell availability is limited, we expect to find a greater use of damaged shells (Hazlett 1966, 1970, Childress 1972, Bertness 1981b). Specimens of *I. sawayai* need heavy shells to survive the strong waves on an exposed sandy beach. The observed pattern of occupation of damaged shells suggests that competition for shell resources made occupation of inadequate shells essential. The natural occurrence of hermit crabs in specific gastropod shells may be a consequence of a preference for such shells and of their relative abundance in the appropriate size range with respect to crab abundance (Conover 1978). We can conclude that the population of *I. sawayai* at La Restinga Beach most probably displays intense shell exchange mechanisms.

One intriguing question arising from this study is, where does *I. sawayai* obtain shells? The crabs may obtain their shells from several different sources: 1) empty shells could be available from El Saco to the West via migratory movements parallel to shore (see Figure 1); 2) the action of waves and currents have an important role in shell transportation from different areas (rocks, mud sediments, and lagoons) such as northwest of La Guardia Bay to the east (see Figure 1), where the most frequently available shells are those more susceptible to physical action (waves and currents); 3) shells may come from older shells deposited in the sediments in the area; and 4) shells may come from intraspecific exchanges. These factors may influence shell availability at La Restinga Beach.

Provenzano (1959) reported a high occupation rate of *Stramonita* gastropod shells by *I. wurdemanni* in Bermuda. Similar results were reported for *I. wurdemanni* in *Stramonita floridana* shells (95.6% occupancy; Caine 1978) in Florida, and in Brazil (49.9% occupancy; Fantucci et al. 2008). The genera *Stramonita*, *Leucozonia*, and *Cymatium* occur over a wide latitudinal range, and their shells are commonly occupied in noticeably high percentages by pagurid and diogenid hermit crabs (Grant and Ulmer 1974, Bertness 1982, Negreiros-Fransozo et al. 1997, Mantelatto and Garcia 1999); thus it is not surprising that they would also be suitable for the hermit crab population studied here.

The presence of live individuals of *Stramonita* and *Leucozonia* in the rocky shore regions of El Saco and La Guardia peripheral to the study area (see Figure 1) can be considered as potential sources of shells appropriate for *I. sawayai*, although proper studies have yet to be done. The genera *Terebra*, *Murex*, and *Melongena* are frequent, but not abundant, in some other reports of shell use by tropical hermit crabs (Provenzano 1959, Fotheringham 1976a, Caine 1978, Bertness 1982), reflecting a similar pattern found in the population studied here. Unfortunately, there are no data or detailed studies available on gastropod species in this area for comparative analysis.

From the results obtained here, we can infer that *I. sawayai* inhabits a wide variety of shell species, but tend to occupy shells of specific species (*L. nassa* and *E. unicinctus*). From published reports it is evident that, in nature, each species of hermit crab preferentially occupies one or a few species of shells, as reported for *Paguristes tortugae* by Mantelatto and Dominciano (2002), *Calcinus tibicen* by Garcia and Mantelatto (2000), *Pagurus brevidactylus* by Mantelatto and Meireles (2004), and *I. sawayai* by Fantucci et al. (2008). These patterns of occupation are related to preferences for specific shells and/or the abundance of appropriate shells in the habitat. However, in the case of *I. sawayai*, the reasons cannot be precisely determined by the study methods used, so further experimental studies dealing with these questions are necessary.

Differences in shell weight are known to encourage use patterns and may affect reproduction of hermit crabs. According to Fotheringham (1976a), shell weight directly affects the amount of energy available for reproduction; crabs carrying heavier shells must shift energy to activities such as locomotion and the search for food. Males of *I. sawayai* were found occupying heavier shells than females and juveniles, probably because of their larger size as well as their numerical dominance or status in the population (Garcia and Mantelatto 2000). Furthermore, larger shells are less subject to displacement by water movements. Larger males preferentially occupied *M. melongena*, probably because of both its ornamental shape (triangular-ovate body and triangular aperture) and medium size. Ovigerous females clearly preferentially occupied *L. nassa* and *E. unicinctus*, but only up to 6.5 mm CSL. Males also showed a higher diversity of shells used (2.27 bit/ind) compared with non-ovigerous females only (1.66 bit/ind). It is possible that males change shells more frequently as a consequence of their faster growth (Mantelatto et al. 2005) as they increase in size and require new and larger shells.

For females, the tendency to occupy larger and heavier shells is related to the space available to contain more eggs (Mantelatto and Garcia 1999). Ovigerous females < 4.7 mm CSL clearly preferred *L. nassa* and *E. unicinctus* shells compared to non-ovigerous females. These observations are supported by Bach et al. (1976), Fotheringham

TABLE 3. *Comparative data on gastropod shell species occupation in hermit crab species of the family Diogenidae from tropical and subtropical zones.*

Hermit species	Number of shell species	Locality	Reference
Calcinus seurati	5	Hawaii, USA	Hazlett (1989)
Calcinus tibicen	21	Randolf Reef, Panama	Bertness (1982)
Calcinus tibicen	7	Ubatuba, Brazil	Mantelatto and Garcia (2000)
Calcinus tubularis	15	Mediterranean Sea, Italy	Pessani et al. (2000a)
Clibanarius albidigitus	7	Golfo Dulce, Costa Rica	Childress (1972)
Clibanarius antillensis	22	Randolf Reef, Panamá	Bertness (1982)
Clibanarius tricolor	18	Florida, USA	Bach et al. (1976)
Clibanarius vittatus	8	Texas, USA	Fotheringham (1976a)
Clibanarius zebra	> 9	Hawaii, USA	Hazlett (1989)
Dardanus insignis	7	Ubatuba, Brazil	Negreiros-Fransozo et al. (1997)
Diogenes nitidimanus	26	Kyushu, Japan	Asakura (1995)
Diogenes pugilator	10	Ligurian Sea, Italy	Pessani et al. (2000b)
Isocheles sawayai	4	Ubatuba, Brazil	Negreiros-Fransozo et al. (1997)
Isocheles sawayai	17	Ubatuba, Brazil	Fantucci (2008)
Isocheles sawayai	26	La Restinga, Venezuela	Present study
Loxopagurus loxochelis	6	Ubatuba, Brazil	Martinelli and Mantelatto (1999)
Paguristes tortugae	21	Ubatuba, Brazil	Mantelatto and Dominciano (2002)
Petrochirus diogenes	12	Ubatuba, Brazil	Bertini and Fransozo (2000)

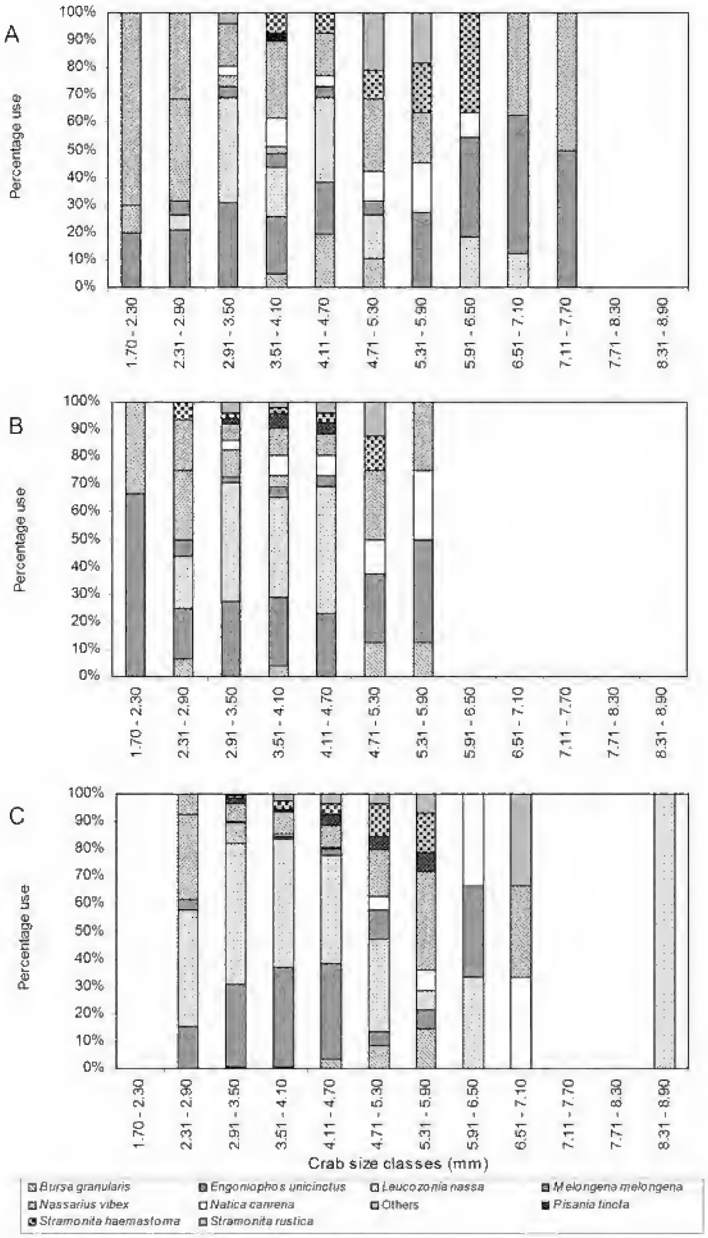

Figure 4.
Percent use of gastropod species shells by Isocheles sawayai in each size class and demographic category.
A. Males.
B. Non-Ovigerous females.
C. Ovigerous females.

(1976b), and Mantelatto and Garcia (1999, 2000). We infer that ovigerous females of *I. sawayai* are more selective regarding shell use in order to 1) reach optimal shell size, 2) use a shell shape that provides a better fit for the crab body shape/size, and 3) allows a better space to carry their broods, as observed in other pagurids of similar size (Dominciano and Mantelatto 2004). In other words, non-ovigerous females may go through a transition period when they look for the appropriate shell to brood their eggs.

In conclusion, we postulate that *I. sawayai* in the La Restinga Beach area shows intense competition for appropriate shells through shell exchange, but concentrated on two gastropod species. The shell use pattern of this species varies between the sexes and the reproductive condition of females. In general, the population inhabits shells of *L. nassa* and *E. unicinctus*, but this occurs mainly for all the ovigerous females. There was a significant size relationship between the hermit crab and its shell, principally with respect to shell width and internal volume.

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Literature Cited

Abbott, R.T. 1974. American Seashells. 2nd ed., Van Nostrand Reinhold Company, Princeton, NJ, USA, 663 p.

Asakura, A. 1995. Sexual differences in life history and resource utilization by the hermit crab. Ecology 76:2295-2313.

Bach, C.B., B. Hazlett, and D. Rittschof. 1976. Effects of interspecific competition on fitness of the hermit crab *Clibanarius tricolor*. Ecology 57:579-586.

Bertini, G. and A. Fransozo. 2000. Patterns of shell utilization in *Petrochirus diogenes* (Decapoda: Anomura: Diogenidae) in the Ubatuba region, São Paulo, Brazil. Journal of Crustacean Biology 20:468-473.

Bertness, M.D. 1981a. The influence of shell type on the growth rates and clutch sizes of hermit crabs. Crustaceana 40:197-205.

Bertness, M.D. 1981b. Seasonality in tropical hermit crab reproduction in the Bay of Panama. Biotropica 13:292-300.

Bertness, M.D. 1982. Shell utilization, predation pressure, and thermal stress in Panamanian hermit crabs: an interoceanic comparison. Journal of Experimental Marine Biology and Ecology 64:159-187.

Caine, E.A. 1978. Habitat adaptations of *Isocheles wurdemanni* Stimpson (Crustacea; Anomura: Diogenidae) and seasonality of occurrences in northwestern Florida. Contributions in Marine Science 21:117-123.

Childress, J.R. 1972. Behavioral ecology and fitness theory in a tropical hermit crab. Ecology 53:960-964.

Conover, M.R. 1978. The importance of various shell characteristics to the shell-selection behavior of hermit crabs. Journal of Experimental Marine Biology and Ecology 32:131-142.

Dominciano, L.C.C. and F.L. Mantelatto. 2004. The influence of shell species and size on the shell selection pattern of *Paguristes tortugae* (Decapoda, Diogenidae) from Anchieta Island (Ubatuba, Brazil). Iheringia, Série Zoologia, 94:425-428.

Fantucci, M.Z., R. Biagi, and F.L. Mantelatto. 2008. Shell occupation by the endemic western Atlantic hermit crab *Isocheles sawayai* (Diogenidae) from Caraguatatuba, Brazil. Brazilian Journal of Biology 68(4):in press

Fantucci, M.Z., R. Biagi, and F.L. Mantelatto. 2007. Record of intersexuality in the western Atlantic hermit crab *Isocheles sawayai* (Anomura, Diogenidae). Journal of the Marine Biological Association of the United Kingdom 2, Biodiversity records 5735: 1-3. Published online (http://www.mba.ac.uk/jmba/pdf/5735.pdf) (accessed 7 January 2008).

Forest, J. and M. De Saint Laurent. 1968. Campagne de la Calypso au large des côtes atlantiques de l'Amérique du Sud (1961-1962). 6. Crustacés Décapodes: Pagurides. Annales de l´Institut océanographique 45:47-169.

Fotheringham, N. 1976a. Population consequences of shell utilization by hermit crabs. Ecology 57:570-578.

Fotheringham, N. 1976b. Hermit crab shells as a limiting resource (Decapoda: Paguridae). Crustaceana 31:193-199.

Garcia, R.B. and F.L. Mantelatto. 2000. Variability of shell occupation by intertidal and infralittoral *Calcinus tibicen* (Anomura, Diogenidae) populations. Nauplius 8:99-105.

Grant, W.C. and K.M. Ulmer. 1974. Shell selection and aggressive behaviour in two sympatric species of hermit crabs. Biological Bulletin 146:32-43.

Guzmán, G. 2004. *Isocheles aequimanus* (Dana, 1852) (Decapoda, Anomura, Paguroidea): revalidación para la carcinofauna chilena. Investigaciones Marinas 32:129-132.

Hazlett, B.A. 1966. The social behaviour of the Paguridae and Diogenidae of Curaçao. Studies on the Fauna of Curaçao and Caribbean Islands 88:1-143.

Hazlett, B.A. 1970. The effect of shell size and weight on the agonistic behavior of a hermit crab. Zeitschrift für Tierpsychologie 27:369-374.

Hazlett, B.A. 1989. Mating success of male hermit crabs in shell generalist and shell specialist species. Behavioural Ecology and Sociobiology 25:119-128.

Imafuku, M. and T. Ando. 1999. Behaviour and morphology of pagurid hermit crab (Decapoda: Anomura) that live in tusk shells (Mollusca, Scaphopoda). Crustaceana 72:129-144.

Mantelatto, F.L. and L.C.C. Dominciano. 2002. Pattern of shell utilization by the hermit crab *Paguristes tortugae* (Diogenidae) from Anchieta Island, Southern Brazil. Scientia Marina 66:265-272.

Mantelatto, F.L. and R.B. Garcia. 1999. Reproductive potential of the hermit crab *Calcinus tibicen* (Anomura) from Ubatuba, São Paulo, Brazil. Journal of Crustacean Biology 19:268-275.

Mantelatto, F.L. and R.B. Garcia. 2000. Shell utilization pattern of the hermit crab *Calcinus tibicen* (Diogenidae) from southern Brazil. Journal of Crustacean Biology 20:460-467.

Mantelatto, F.L. and A.L. Meireles. 2004. The importance of shell occupation and shell availability in the hermit crab *Pagurus brevidactylus* (Stimpson, 1859) (Paguridae) population from Southern Atlantic. Bulletin of Marine Science 75:27-35.

Mantelatto, F.L., R.A. Christofoletti, and W.C. Valenti. 2005. Population structure and growth of the hermit crab *Pagurus brevidactylus* (Anomura: Paguridae) from the northern coast of São Paulo, Brazil. Journal of the Marine Biological Association of the United Kingdom 85:127-128.

Mantelatto, F.L., R. Robles, R. Biagi, and D.L. Felder. 2006. Molecular analysis of the taxonomic and distributional status for the hermit crab genera *Loxopagurus* Forest, 1964, and *Isocheles* Stimpson, 1858 (Decapoda, Anomura, Diogenidae). Zoosystema 28:495-506.

Margalef, R. 1974. Ecología. Omega, Barcelona, Spain. 951 p.

Martinelli, J. and F.L. Mantelatto. 1999. Shell utilization by the hermit crab *Loxopagurus loxochelis* (Diogenidae) in Ubatuba Bay, Brazil. In: F.R. Schram and J.C. Vaupel Klein, eds. Crustaceans and the Biodiversity Crisis. Proceedings of the Fourth International Crustacean Congress, Vol. 1, Koninklijke Brill NV, Leiden, The Netherlands, p. 719-731.

Meireles, A.L. and F.L. Mantelatto. 2005. Shell use by *Pagurus brevidactylus* (Anomura, Paguridae): a comparison between laboratory and field conditions. Acta Zoologica Sinica 51:813-820.

Morris, P.A. 1975. A Field Guide to Shells of the Atlantic and Gulf Coasts and the West Indies. 3rd ed. Houghton Mifflin Company, Boston, MA., USA, 330 p.

Negreiros-Fransozo, M.L., A. Fransozo, F.L. Mantelatto, M.A.A. Pinheiro, and S. Santos. 1997. Anomuran species (Crustacea: Decapoda) and their ecological distribution at Fortaleza Bay sublittoral, Ubatuba, São Paulo, Brazil. Iheringia, Série Zoologia 83:181-186.

Negreiros-Fransozo, M.L. and N.J. Hebling. 1983. Desenvolvimento pós-embrionário de *Isocheles sawayai* Forest & Saint Laurent, 1967 (Decapoda, Diogenidae) em laboratório. Papéis Avulsos de Zoologia 35:41-53.

Nucci, P.R. and G.A.S. Melo. 2000. Range extensions for eight species of western Atlantic hermit crabs (Crustacea, Paguroidea). Nauplius 8:141-147.

Pessani, D., M.T. Damiano, G. Maiorana, and T. Tirelli. 2000a. The biology of the hermit crab *Calcinus tubularis* (Decapoda, Diogenidae) in nature and in the laboratory. In: F.R. Schram and J.C. Vaupel Klein, eds. Crustacean Issues 12.

Crustaceans and the Biodiversity Crisis. Proceedings of the Fourth International Crustacean Congress, Vol. 2. Brill, Leiden, Netherlands, p. 377-383.

Pessani, D., M. Dappiano, T. Tirelli, and C. Cerrano. 2000b. The biology of a population of the hermit crab *Diogenes pugilator* (Decapoda, Diogenidae) in the Ligurian Sea. In: F.R. Schram and J.C. Vaupel Klein, eds. Crustacean Issues 12. Crustaceans and the Biodiversity Crisis. Proceedings of the Fourth International Crustacean Congress, Volume 2, Brill, Leiden, Netherlands, p. 323-329.

Pinheiro, M.A.A., A. Fransozo, and M.L. Negreiros-Fransozo. 1993. Seleção e relação com a concha em *Isocheles sawayai* Forest and Saint Laurent, 1967 (Crustacea: Anomura: Diogenidae). Arquivos de Biologia e Tecnologia 36:745-752.

Provenzano, A.J. Jr. 1959. The shallow water hermit crabs of Florida. Bulletin of Marine Science of the Gulf and Caribbean 9:349-420.

Reese, E.S. 1969. Behavioural adaptations of intertidal hermit crabs. American Zoologist 9:343-355.

Scully, E.P. 1979. The effects of shell availability on intraspecific competition in experimental populations of the hermit crab, *Pagurus longicarpus* Say. Journal of Experimental Marine Biology and Ecology 71:221-236.

Vance, R.R. 1972. The role of shell adequacy in behavioral interactions involving hermit crabs. Ecology 53:1075-1083.

Warmke, G.L. and R.T. Abbott. 1962. Caribbean Sea Shells. Livingston Publishing Company, Narberth, PA, USA, 450p.

Zar, J.H. 1996. Biostatistical Analysis. Prentice-Hall, Upper Saddle River, NJ, USA, 907p.

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SEASONAL PATTERNS OF JUVENILE FISH ABUNDANCE IN SEAGRASS MEADOWS IN TEAGUE BAY BANK BARRIER REEF LAGOON, ST. CROIX, U.S. VIRGIN ISLANDS

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ABSTRACT: Considerable knowledge has been gained regarding fish use of nearshore habitats such as seagrass meadows or mangrove lagoons in the Caribbean (e.g., evaluation of nursery value, trophic linkages). However, few studies have been conducted on fish recruitment to seagrass habitat around the Caribbean. Juvenile reef fish in seagrass meadows at Teague Bay, St Croix, U.S. Virgin Islands were surveyed from October 1998 through September 1999 using a visual census technique. Grunts (Haemulidae) were the most abundant juveniles observed (60% of all fish), followed by wrasses (Labridae, 20%) and parrotfishes (Scaridae, 13%). French grunt, *Haemulon flavolineatum*, were the most numerous species (59.5% of all fish), followed by slippery dick, *Halichoeres bivittatus* (18.5%), and bucktooth parrotfish, *Sparisoma radians* (10.4%). Most numerically abundant fish species demonstrated peaks in recruitment during late summer and fall. Our results imply that the functioning of seagrass beds incorporates strong seasonal patterns of small-fish abundance that need to be accommodated in any study wishing to understand their importance to fisheries.

INTRODUCTION

Seasonal patterns of recruitment have been studied extensively in coral reef habitats at various locations, such as Great Barrier Reef, French Polynesia, Hawaii, and the Caribbean (Williams and Sale 1981, Eckert 1984, Walsh 1987, Doherty 1991, Dufour 1993, Casselle and Warner 1996, Planes 1997, Robertson and Kauffman 1998). However, there have been no studies on seasonal fish recruitment patterns within different coastal habitats such as seagrass beds, mangroves, and backreefs, despite the widely accepted view of these habitats as juvenile nursery grounds (Nagelkerken et al. 2000a, 2000b, Cocheret et al. 2002, Mumby et al. 2004). Ogden and Gladfelter (1983) claim these nearshore habitats act as nurseries for three main reasons: 1) they are located away from the heavy predation pressure characteristic of coral reefs, 2) they offer protection to small fishes due to the structural complexity of masses of leaves and roots, and 3) they provide a rich food supply based on plant detritus and associated microorganisms and small invertebrates. In addition, most studies of nearshore tropical fish habitat use (Nagelkerken et al. 2000a, 2000b, 2001, Cocheret et al. 2002, Halpern 2004, Mumby 2004, Chittaro et al. 2005) were conducted in short periods of time (1 to 4 months) without taking into consideration the seasonality of these species. Thus, it is critical to investigate seasonality of fish recruitment in seagrass beds in order to refine our knowledge of coastal fish habitat use.

Because critical seagrass habitats are generally close to shore, they are susceptible to anthropogenic disturbances such as storm-water and pollutant runoff and spills and mechanical damage by boats. With growing fears that stock restoration efforts are being compromised more by habitat loss from coastal development and by pollution than by overexploitation, conservation of habitats (such as seagrass meadows) is becoming an important part of fisheries management.

In order to support informed decisions for the sustainable management of marine fish and their habitats, there is a vital need for more documentation on the seasonality of habitat use by small juveniles. The goal of this study was to document temporal recruitment patterns in the fish assemblages in seagrass meadows in the U.S. Virgin Islands. This study was designed to answer the following questions: (1) Are there significant variations (order of magnitude) in recruitment patterns among the most abundant seagrass fish species? (2) Are there clear seasonal patterns in recruitment among seagrass-associated fish species?

Materials and Methods

The three embayments sampled in this study (Cottongarden Bay, Teague Bay, and Yellowcliff Bay) are part the Teague Bay bank-barrier reef system that extends from Pull Point to Lamb Point on the Northeast coast of St. Croix (Figure 1). This lagoon is described in Mateo and Tobias (2001). All seagrass meadows were found at similar depths (0.5 m to 3 m), and the vegetation within beds was dominated by turtlegrass *Thalassia testudinum* and manatee grass *Syringodium filiforme* with percent seagrass coverage at about 80%. From October 1998 through September 1999, fish recruits (recently settled post-larvae and juveniles) were counted along 50 m x 2 m strip transects (Fowler et al. 1992). For each bay, a 20 m x 20 m grid pattern was laid over a nautical chart. Grid intersecting points were labeled with consecutive numbers and were the bases for selecting transect starting points for each embayment. Ten randomly selected starting points were surveyed per month for all three embayments, based

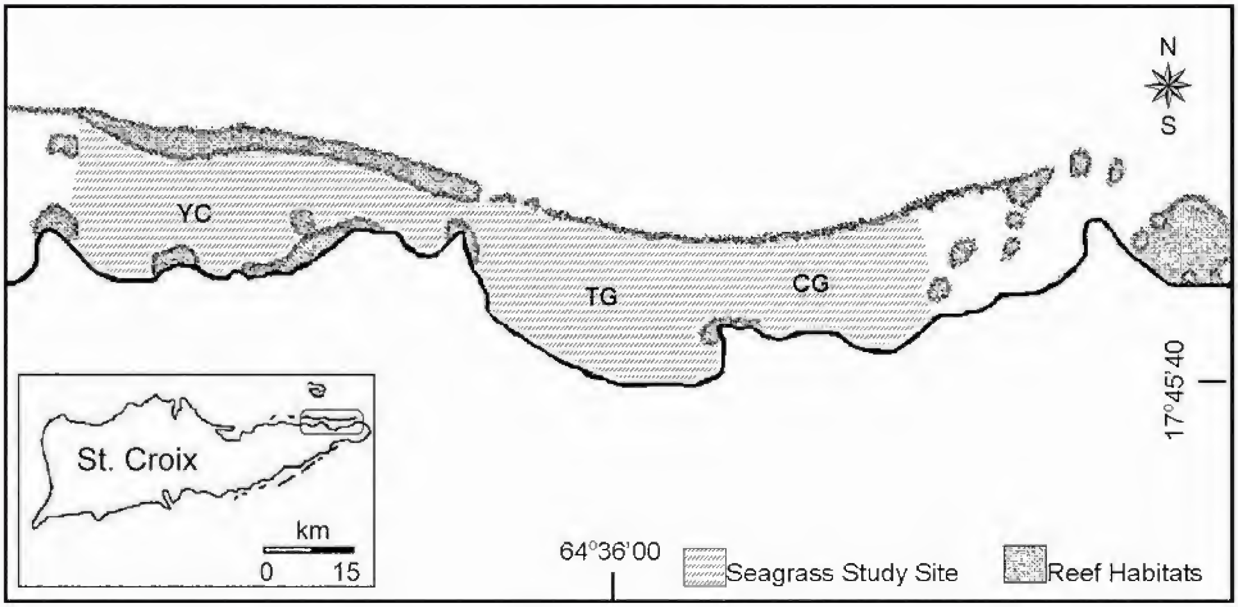

Figure 1.
Location of seagrass meadows study site at Yellowcliff Bay (YC), Teague Bay (TG), and Cottongarden Bay (CG) at the Northeast coast of St. Croix, USVI.

on a preliminary fish census that used cumulative species/transect counts Rogers et al. (1994). At each of the starting points, a single 50-m transect line (marked at 1 cm intervals) was laid out on a randomly selected compass bearing for each transect. On each transect, 100 m^2 were visually surveyed for fish by two divers swimming parallel but on opposite sides of the transect in a 1 m x 50 m belt transect.

At each transect site, a fish census and a benthic survey were done. Each diver recorded fish species and estimated the size classes (<5 cm, 5-10 cm, and > 10 cm total length [TL]) of individuals for each species. For most species, juveniles ≤ 5 cm were recorded as recruits. For smaller species, such as wrasses, grunts, and damselfishes, juveniles ≤ 3 cm were considered recruits. Only juvenile recruits were considered for analysis in the remainder of this study. Identification of grunt recruits was verified by both divers during each dive using an underwater guide of early life history of grunts taken from Lindeman (1997). To minimize the potential bias of counting the same individual twice along the belt transect, divers conferred with each other using hand signals to make sure fish were counted only once (Eberhardt 1978), and divers were trained to maintain constant swimming speed along the transect and to not count fish that entered the census area after the visual census had started (Samoylis and Carlos 2000).

Statistical Analysis

Prior to conducting data analyses, fish density estimates from both divers were checked for independence with a Pearson product-moment correlation coefficient (r) (Zar 1984). If uncorrelated, the paired transects could be considered independent samples. We considered $r < 0.50$ to indicate independence. Correlation between paired divers was low ($r = 0.41$, $p = 0.243$, $n = 360$), and we interpreted the data generated from the two divers as separate and independent census data sets. Data were standardized by month by pooling belt transects from all three embayments by habitat type. This allowed for equal sample size ($n = 24$) for the one year study.

The assumption of homogeneity of variance was tested prior to each analysis using the Levene Median test (Zar 1984) for data on number of fish per transect and density of the most abundant species. If this assumption was violated, we log (x+1)-transformed data to satisfy assumptions of homogeneity of the variances (non-transformed data were used in graphs for clarity). Monthly variation in density of the most abundant species recorded on transects were examined with a two-way ANOVA (Sokal and Rohlf 1981). If the overall F-value was significant, Tukey's pair-wise multiple comparison procedure was used to compare mean values.

Results

A total of 8,243 juveniles of 23 species were counted during the study (Table 1). Grunts (Haemulidae) were the most abundant family comprising 60.1% of all juveniles observed. Wrasses (Labridae) were the second most abundant family with 19.4% of the total, followed by parrotfishes (Scaridae, 13.3%). Eight other families comprised the remaining 7.2% of juveniles observed. Of 23 species observed, the French grunt, *Haemulon flavolineatum*, was overwhelmingly dominant, accounting for 59.5% of all recruits, followed by slippery dick, *Halichoeres bivittatus* (18.5%), and bucktooth parrotfish, *Sparisoma radians* (10.4%) (Table 1).

Significant differences in fish recruit density were found among species ($F_{7,11} = 23.175$, $p < 0.001$) and month ($F_{7,11} = 20.737$, $p < 0.001$) for all taxa. Significant interactions among

species and month ($F_{7,11}$= 1.791, p < 0.003) were also found in this study. Recruit densities of H. *flavolineatum*, *H. bivittatus*, and S. *radians* were significantly higher than those for the remaining species (Tukey test, p < 0.001). There were also significant differences in small fish abundance of *H. flavolineatum*, *H. bivittatus*, *S. radians*, *Ocyurus chrysurus*, *Scarus iseri*, and *Acanthurus chirurgus* among months (Tukey test, p < 0.001).

Major recruitment peaks for *H. flavolineatum* were observed in November 1998 and July 1999 (Figure 2a). Because sampling was only conducted for 12 months, it is uncertain whether these peaks indicate annual or semi-annual pulses. The second most abundant species (*H. bivittatus*) exhibited abundance peaks in October 1998 and September 1999 and lower recruitment during other months (Figure 2b), indicating a prolonged recruitment period with a peak during autumn. Bucktooth parrotfish, *S. radians*, clearly exhibited bimodal recruitment, with peaks in October 1998, May 1999, and September 1999 (Figure 2c). Doctorfish, *A. chirurgus*, exhibited continuous recruitment from April to November and no recruitment from December to March (Figure 2d). Yellowtail snapper, *O. chrysurus*, recruits exhibited an annual peak in August and September of 1999 (Figure 2e). A similar pattern was observed for beaugregory, *Stegastes leucostictus*, with continuous recruitment from April to January and no recruits seen from February to March (Figure 2f). Blackear wrasse, *Halichoeres poeyi*, (Figure 2g) followed the same recruitment pattern as *H. bivittaus*. The striped parrotfish, *S. iseri*, showed a large peak in October 1998 (Figure 2h).

TABLE 1. *Abundance of fish recruits on seagrass meadows in Teague Bay, St. Croix, U.S. Virgin Islands, October 1998 to September 1999 using visual census. Total area surveyed was 36,000 m².*

Family and Species	Total Recruits	Percent of Total
Synodontidae		
Synodus foetens	1	0.01
Lutjanidae		
Ocyurus chrysurus	214	2.60
Lutjanus synagris	11	0.13
Lutjanus mahogoni	7	0.09
Haemulidae		
Haemulon flavolineatum	4,901	59.51
Haemulon plumierii	56	0.68
Mullidae		
Pseudupeneus maculatus	29	0.35
Chaetodontidae		
Chaetodon capistratus	30	0.36
Pomacentridae		
Stegastes leucostictus	123	1.49
Stegastes partitus	6	0.07
Labridae		
Halichoeres bivittatus	1,524	18.51
Halichoeres poeyi	72	0.87
Xyricthys martinicensis	7	0.09
Doratonotus megalepsis	4	0.05
Scaridae		
Sparisoma radians	860	10.44
Scarus iseri	155	1.88
	62	0.75
Acanthuridae		
Acanthurus chirurgus	122	1.48
Acanthurus bahianus	22	0.27
Monacanthidae		
Monacanthus ciliatus	9	0.11
Tetradontidae		
Canthigaster rostrata	11	0.13
Sphoeroides spengleri	7	0.09
Sphoeroides testudineum	2	0.02
TOTAL	**8,235**	**100.00**

Discussion

Although conclusions cannot be drawn from only one year of data, some of the common seagrass fishes in St. Croix appeared to show seasonal variation in recruitment pulses. Within our eight most abundant species we found that *H. flavolineatum*, *H. bivittatus*, *S. radians*, *O. chrysurus*, *S. iseri*, and *H. poeyi* had major recruitment pulses from late summer to late fall. In the Caribbean region, studies focusing on entire reef fish assemblages (not just those in seagrass) have documented seasonal recruitment, primarily during spring through fall. In Barbados, Tupper and Hunte (1994) found that assemblage-wide recruitment was high between May and November and low between December and April. Luckhurst and Luckhurst (1977) reported semi-annual recruitment pulses, primarily in the spring and fall, for sixteen species within seven families in the Netherlands Antilles. Beets (1997) found abundance peaks of fish recruits on artificial reefs in St. Thomas USVI in April and June. Finally, late spring-summer peaks in recruitment were documented for four of the five most abundant families (Gobiidae, Labridae, Haemulidae, Pomacentridae) in a fringing reef in St. John USVI (Miller et al. 2001).

The recruitment patterns exhibited by *H. bivittatus*, *H. poeyi*, *A. chirurgus*, and *S. iseri* contrasted with those found in other studies in the Caribbean. Luckhurst and Luckhurst (1977) reported year-round labrid recruitment with spring pulses in the Netherlands Antilles, whereas in our study *H. bivittatus* abundance peaks occurred during September and October. Adams and Ebersole (2002) reported recruitment peaks in June and February for acanthurid species on lagoonal patch reefs in St. Croix, while in

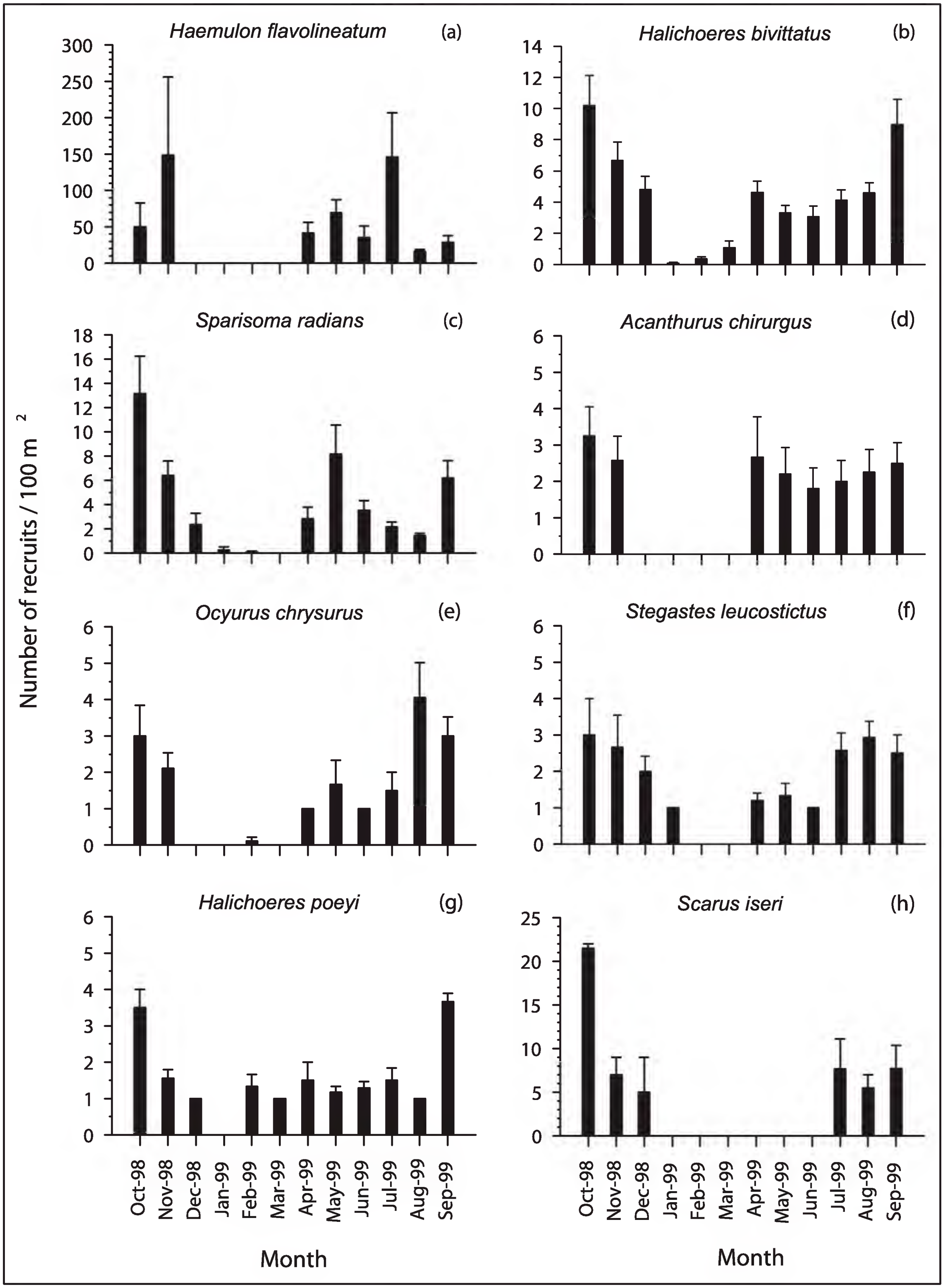

Figure 2.
Mean monthly abundance (± standard error) of dominant recruit species observed using visual census on seagrass meadows in Teague Bay from October 1998 to September 1999. n=10 transects per month (area of each transect = 100m^2).

our study we found comparable densities of *A. chirurgus* recruits from spring through fall. For *S. iseri*, Miller et al. (2001) found recruitment peaks in summer, while we found more *S. iseri* recruiting from late summer to late fall, with a major recruitment peak during the month of October.

Signifficant geographical variation in seasonality of juvenile abundance seems likely to occur throughout the Caribbean (Victor 1991, Robertson and Kauffman 1998). Distinct intraspecific geographical variation in spawning seasonal ity has been reported within the Caribbean, with a tendency towards less seasonality in the more tropical parts of that region in some species and spawning peaks at different times of the year at different locations in others (Victor 1991, Robertson and Kauffman 1998). Reef fish species differ in the extent to which their recruitment seasonality varies in different parts of the Caribbean (Robertson and Kauffman 1998): for example, while four *Stegastes* species have somewhat strong late summer peaks of recruitment in the Northwest Caribbean (Booth and Beretta 1994, Mcghee 1995), they have weaker seasonality in the Southwest Caribbean, with their recruitment peaks at least 6 months earlier in the year (Robertson 1990). Furthermore, pomacentrid species may have different seasonal recruitment peaks at different locations on a single island in the same year (Booth and Beretta 1994, Caselle and Warner 1996). Damselfish recruitment occurs during September in Puerto Rico (McGehee 1995), year-round with fall pulses in the Netherlands Antilles (Luckhurst and Luckhurst 1977), and from June to September in Barbados (peak for *Stegastes partitus*; Tupper and Hunte 1994). In our study, damselfish exhibited summer/fall recruitment pulses. Other species appear to have consistent recruitment patterns throughout the Caribbean. For example, *H. flavolineatum* has been found to recruit throughout the year in St. Croix (McFarland et al. 1985, Shulman and Ogden 1987), with reports of recruitment peaks in summer and fall throughout the Caribbean (Miller et al. 2001, Appeldoorn et al. 1997, this study). For *O. chrysurus*, recruitment peaks found in the present study (during August-October) were similar to those found by Watson et al. (2002) in seagrass habitats in Tortola.

In recent years, considerable knowledge has been gained regarding fish use of nearshore habitats such as seagrass meadows or mangrove lagoons in the Caribbean (e.g., evaluation of nursery value or trophic linkages) (Nagelkerken et al. 2000a, 2000b, Cocheret et al. 2003, Mumby et al. 2004, Chittaro et al. 2005); however, few studies have been conducted on fish recruitment to seagrass habitat around the Caribbean. This is a component that is often overlooked in studies characterizing the nursery roles of seagrass and mangroves for tropical fishes in the Caribbean (Nagelkerken et al. 2000a, 2000b, 2001, Cocheret et al. 2002, Mumby et al. 2004, Chittaro et al. 2005).

Our study has demonstrated differences in recruitment intensity among species and months within a year of study. We recognize that this is a short-term study. It may or may not be indicative of typical recruitment patterns but provides valuable comparative information on recruitment from the Caribbean region. The observed variability in recruitment indicates the need for frequent sampling. Our results imply that the functioning of any seagrass bed incorporates strong seasonal patterns which need to be accommodated in any study wishing to understand their importance to fisheries.

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Literature Cited

Adams, A. and J. Ebersole. 2002. Use of lagoon habitats and back reef by coral reef fishes. Marine Ecology Progress Series 228:213-226.

Appeldoorn, R.S., C.W. Recksiek, R.L. Hill, F.E. Pagan, and G.D. Dennis. 1997. Marine protected areas and reef fish movements: the role of habitat in controlling ontogenetic migration. Proceedings Eighth International Coral Reef Symposium 2:1917-1922.

Beets, J. 1997. Effects of a predatory fish on the recruitment and abundance of Caribbean coral reef fishes. Marine Ecology Progress Series 148:11-21.

Caselle, J.E. and R.R. Warner. 1996. Variability in recruitment of coral reef fishes: the importance of habitat at two spatial scales. Ecology 77:2488-2504.

Chittaro, P.M., P. Usseglio, and P.F. Sale. 2005. Variation in fish density, assemblage composition and relative rates of predation among mangrove, seagrass and coral reef habitats. Environmental Biology of Fishes 72:175-187.

Cocheret de la Morinière, E., B.J.A. Pollux, I. Nagelkerken, and G. van der Velde. 2002. Post-settlement life cycle migration patterns and habitat preference of coral reef fish that use seagrass and mangrove habitats as nurseries. Estuarine, Coastal and Shelf Science 55:309-321.

Cocheret de la Morinière, E., B.J.A. Pollux, I. Nagelkerken, M.A. Hemminga, A.L. Huiskes, and G. van der Velde. 2003. Ontogenetic dietary changes of coral reef fishes in the mangrove-seagrass-reef continuum: stable isotopes and gut contents analysis. Marine Ecology Progress Series 246:279-289.

Doherty, P.J. 1991. Spatial and temporal patterns in recruitment. In: Sale, P.F. ed. The Ecology of Fishes on Coral Reefs. Academic Press Inc., San Diego, CA, USA, p. 261-293.

Dufour V. and R. Galzin. 1993. Colonization patterns of reef fish larvae to the lagoon at Moorea Island, French Polynesia. Marine Ecology Progress Series 102:143-152.

Eberhardt, L.L. 1978. Transect methods for population studies. Journal of Wildlife Management. 42:1-31.

Eckert, G.J. 1984. Annual and spatial variation in recruitment of labroid fishes among seven reefs in the Capricorn / Bunker Groups, Great Barrier Reef. Marine Biology 78:123-127.

Fowler, A.J., P.J. Doherty, and D.McB. Williams. 1992. Multiscale analysis of recruitment of a coral reef fish on the Great Barrier Reef. Marine Ecology Progress Series 82:131-141.

Halpern, B.S. 2004. Are mangroves a limiting resource for two coral reef fishes? Marine Ecology Progress Series 272:93-98.

Hunt von Herbing, I. and W. Hunte. 1991. Spawning and recruitment of the bluehead wrasse *Thalassoma bifasciatum* in Barbados, West Indies. Marine Ecology Progress Series 72:49-58.

Lindeman, K.C. 1997. Development of grunts and snappers of southeast Florida; crossshelf distributions and effects of beach management altenatives. Ph.D. dissertation. University of Miami, Miami, FL, USA, 420p

Luckhurst, B.E. and K. Luckhurst. 1977. Recruitment patterns of coral reef fishes on the fringing reef of Curaçao, Netherlands Antilles. Canadian Journal Zoology 55:681-689.

Mateo, I. and W.J. Tobias. 2001. Distribution of shallow water coral reef fishes on the northeast coast of St. Croix, USVI. Caribbean Journal of Sciences 37:210-226.

McFarland, W.N., E.B. Brothers, J.C. Ogden, M.J. Shulman, E.L., Berminghamand, and N.M. Kotchian-Prentiss. 1985. Recruitment patterns in young french grunts, *Haemulon flavolineatum* (family Haemulidae) at St. Croix, U.S.V.I. Fishery Bulletin 83:413-426.

McGehee, A.M. 1995. Juvenile settlement, survivorship and in situ growth rates of four species of Caribbean damselfishes in the genus *Stegastes*. Environmental Biology of Fishes 44:393-401.

Miller J., J. Beets, and C. Rogers. 2001. Temporal patterns of fish recruitment on a fringing coral reef in Virgin Islands National Park, St. John, U.S. Virgin Islands. Bulletin of Marine Sciences 69:567-579.

Mumby, P.J., A.J. Edwards, J.E. Arias-Gonzalez, K.C. Lindeman, P.G. Blackwell, A. Gall, M.I. Gorczynska, A.R. Harborne, C.L. Pescod, H. Renken, C.C.C. Wabnitz, and G. Llewellyn. 2004. Mangrove enhances the biomass of coral reefs fish communities in the Caribbean. Nature 427:533-536.

Ogden, J.C. and E.H. Gladfelter. 1983. Coral Reefs, seagrass beds, and mangroves: Their interaction in the coastal zones of the Caribbean. UNESCO Reports in Marine Science 23, 133 p.

Nagelkerken, I., M. Dorenbosch, W.C.E.P. Verbek, E. Cocheret de la Morinie`re, and G. Van der Velde. 2000a. Importance of shallow-water biotopes of a Caribbean bay for juvenile coral reef fishes: patterns in biotope association, community structure and spatial distribution. Marine Ecology Progress Series 202:175-192.

Nagelkerken I, G. van der Velde, M.W. Gorissen, G.J. Meijer, T. van't Hof, and C. den Hartog. 2000b. Importance of mangroves, seagrass beds and the shallow coral reef as a nursery for important coral reef fishes, using a visual census technique. Estuarine, Coastal and Shelf Science 51:31-44.

Nagelkerken I., S. Kleijnen, T. Klop, R.A.C.J. van den Brand, E. Cocheret de la Morinière, and G. van der Velde. 2001. Dependence of Caribbean reef fishes on mangroves and seagrass beds as nursery habitats: a comparison of fish faunas between bays with and without mangroves/seagrass beds. Marine Ecology Progress Series 214:225-235.

Robertson, D.R. 1990. Differences in the seasonalities of spawning and recruitment of some small neotropical reef fishes. Journal of Experimental Marine Biology and Ecology. 144:49-62.

Robertson, D.R. and K.W. Kauffman. 1998. Assessing early recruitment dynamics and its demographic consequences among tropical reef fishes: Accommodating variation in recruitment seasonality and longevity. Australian Journal Ecology 23:226-233.

Rogers, C. S., G. Garrison, R. Grober, Z. Hillis, and M. Franke. 1994. Coral Reef Monitoring Manual for The Caribbean and Western Atlantic. Technical Report, National Park Service, Virgin Islands National Park , St John, USVI, 42 p.

Samoilys, M. and G. Carlos. 2000. Determining methods of underwater visual census for estimating the abundance of coral reef fishes. Environmental Biology of Fishes 57:287-309.

Shulman, M.J. and J.C. Ogden. 1987. What controls tropical reef fish populations: recruitment or benthic mortality? An example in the Caribbean reef fish *Haemulon flavolineatum*. Marine Ecology Progress Series 39:233-242.

Sokal, R. R. and F. J. Rohlf. 1981. Biometry. W.H. Freeman and Company, New York, NY, USA, 959 p.

Tupper, M. and W. Hunte. 1994. Recruitment dynamics of coral reef fishes in Barbados. Marine Ecology Progress Series 108:225-235.

Victor, B.C. 1991. Settlement strategies and biogeography of reef fishes. In: Sale, P.F. ed. The Ecology of Fishes on Coral Reefs. Academic Press Inc., San Diego, CA, USA, p. 231-292.

Walsh, W.J. 1987. Patterns of spawning and recruitment for Hawaiian reef fishes. Environmental Biology 18:257-278.

Watson, M., J. Munro, and F. Gell. 2002. Settlement, movement and early juvenile mortality of the yellowtail snapper *Ocyurus chrysurus*. Marine Ecology Progress Series 237:247-256.

Williams, D.McB and P.F. Sale. 1981. Spatial and temporal patterns of recruitment of juvenile coral reef fishes to coral habitats within One Tree Lagoon, Great Barrier Reef. Marine Biology 64:245-253.

Zar, J.H. 1984. Biostatistical Analysis, 2nd ed. Prentice Hall, Upper Saddle River, NJ, USA, 718p.

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CARPOAPSEUDES HEARDI N. SP. (TANAIDACEA: APSEUDOMORPHA) FROM CARIBBEAN WATERS NEAR TOBAGO

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Abstract: *Carpoapseudes heardi* n. sp. is described from samples collected from depths of 421 and 537 m off Tobago and is the first Caribbean record for the genus. The new species bears a resemblance to *Carpoapseudes serratospinosus* Lang, 1968 and other related species in the shortened dactylus unguis combination of pereopod 1, but has parallel eyespines like *Carpoapseudes bacescui* Guţu, 1975 and *Carpoapseudes simplicirostris* (Norman and Stebbing, 1886). Other diagnostic characters include pereopods 2 and 3 with basal spurs, labrum with paired lobes, labial palp with two terminal setae, maxillipedal bases with outer crenulations, and pleopods with 1-articled rami. It was found to lack an epistomal spine and has an unusual form of the third pereopod short propodal spine.

Resumo: *Carpoapseudes heardi* sp. nov. é descrito de amostras coletadas em profundidades entre 421 e 537 m ao largo de Tobago, o primeiro registro do gênero no Caribe. A nova espécie se assemelha a *Carpoapseudes serratospinosus* Lang, 1968 e outras espécies no encurtamento da combinação dátilo-unguis no pereópodo 1, mas possui lobos oculares paralelos como *Carpoapseudes bacescui* Guţu, 1975 and *Carpoapseudes simplicirostris* (Norman e Stebbing, 1886). Outros caracteres diagnósticos incluem pereópodos 2 e 3 com esporas basais, labrum com lobos pareados, palpo labial com duas cerdas terminais, base do maxilípede com crenulações externas e pleópodos com ramo uniarticulado. É diferenciada pela ausência do espinho do epistoma e possui um curto e incomum espinho no própodo do terceiro pereópodo.

Introduction

Recent deep-sea sampling efforts in the Caribbean Sea near Trinidad and Tobago conducted by Continental Shelf Associates (CSA) International Inc. under contract to Petro-Canada Trinidad and Tobago Ltd. have revealed a rich assemblage of tanaidaceans and other peracarideans. Samples were collected by CSA in the wet (November-December 2006) and dry (May-June 2006) seasons, and both collections included specimens of a new species of *Carpoapseudes*. Several other new apseudomorphs and tanaidomorphs were also found which will be the subject of future reports. Recent studies have shown that the ranges of some Caribbean deep-sea tanaids overlap with those in the Gulf of Mexico (GOM; T.J. Hansknecht, unpublished data), although no published records of the genus *Carpoapseudes* have been reported for GOM waters (Larsen 2005) or the Caribbean Sea. The nearest reported species (Larsen 1999) in the genus is *Carpoapseudes prospectnes* Larsen, 1999 from the South Atlantic off Brazil.

Guţu (1972) originally proposed the subfamily Leiopinae to include the genera *Carpoapseudes* and *Leiopus* Beddard, 1886 that have a first pereopod stronger than the following ones and a maxillipedal endite bearing a specialized inner distal seta with a leaf-shaped form. The genus *Carpoapseudes* Lang, 1968 consists of a group of 17 large, deep-sea apseudomorphs, from all major oceans. These tanaids are characterized by a lengthening of the carpus on pereopod 1 as compared to the merus (Guţu, 1980). In contrast, Băcescu (1982) reported *Carpoapseudes curticarpus* from the NE Atlantic which is the first known species with a pereopod 1 carpus shorter than the merus, and *Carpoapseudes prospectnes* Larsen, 1999 as the first member of the genus with the carpus of pereopod 1 equal to the merus in length (Larsen 1999). Members of this genus also share a characteristic male cheliped morphology with a large triangular tooth on the propodal fixed finger. Guţu (1996) later presented a key to the thirteen known species of *Carpoapseudes* based, in part, on a comparison of the rostral lengths, pereopod 2 carpal length relative to that of the merus, and the length of the exopod relative to that of the basis of the cheliped. Other key characters included the number of antennule and antenna articles of the flagella and the presence or absence of branchial spines.

Other members of the genus have been described worldwide. For example, *Carpoapseudes caraspinosus* Dojiri and Sieg, 1997 was described from California and was the second species of the genus to be found with lateral spines on the carapace, and *Carpoapseudes spinigena* Bamber, 2007 and *Carpoapseudes varindex* Bamber, 2007 have recently been described from the Kurile-Kamchatka and the Japan trenches. The purpose of this study was to report the first occurrence of this genus in the Caribbean Sea and describe a new species of *Carpoapseudes*.

Materials and Methods

Shipboard samples were collected by CSA using a 0.35-m^2 box corer, and infaunal samples were preserved in 10% formalin, stained with Rose Bengal, sorted, and then

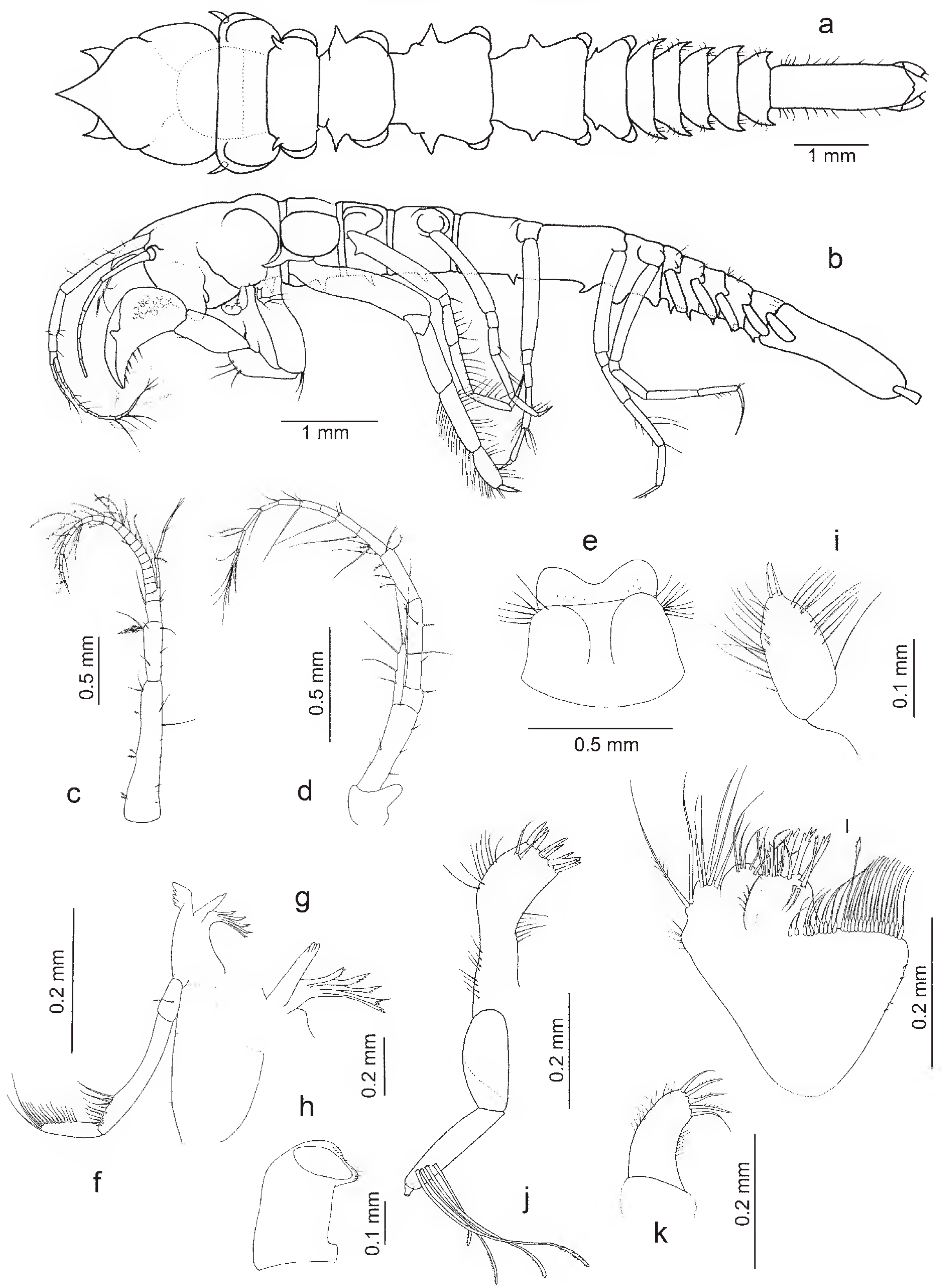

Figure 1.
Carpoapseudes heardi n. sp. (a) ovigerous female (11.2 mm) paratype dorsal view, (b) male holotype (8.3 mm) lateral view, (c) antennule, (d) antenna, (e) labrum, (f) left mandible, (g) same, spine row enlarged, (h) molar process, (i) labium, (j) maxillule palp and outer endite, (k) maxillule inner endite, and (l) maxilla.

stored in 70 % isopropyl alcohol. Samples were sorted and identified by Barry Vittor and Associates (BVA), Mobile, Alabama. Slide mounted specimens were studied with a Leica MZ6 and illustrated with a Leitz camera luci-

da-equipped Nikon Optiphot. All measurements are given in millimeters. Types are deposited in the United States National Museum (USNM) in Washington, D.C. and additional material is retained at BVA. The terminology of Watling (1989) and Larsen (2003) is followed, with the exception that spiniform setae on the pereopod articles are called spines. The terms parallel and divergent are proposed for the orientation of the eye spines. Anteriorly directed eye spines, in relation to the carapace, are referred to here as parallel, whereas outward directed spines are divergent.

SYSTEMATIC ACCOUNT

Suborder Apseudomorpha Sieg, 1968

Family Apseudidae Leach, 1814

Subfamily Leviapseudinae Sieg, 1980

Genus *Carpoapseudes* Lang, 1968

Diagnosis (modified from Lang (1968), Guţu (1996) and Larsen (1999))

Carapace with rostrum and eye lobes bearing a terminal spine. Pereonites 3-5 with lateral spines; pereonite 6 shorter than pereonites 3-5, trapezoidal in shape. Pleotelson cylindrical and long. Antenna with 5-articled peduncle; squama well-developed. Mandible with 3-articled palp. Maxillule with 2-articled palp. Endite of maxilliped with a leaf-like spine. Chela and pereopod 1 with exopodites. Pereopod 1 coxa with spine. Pereopod 1 with carpus and propodus, dorsal and ventral margins lined with numerous finely attenuate setae. Chelipeds sexually dimorphic, with triangular tooth on fixed finger in males. Pleopods well-developed, five pairs.

Described species included in genus: *Carpoapseudes auritochelis* Kudinova–Pasternak, 1975; *C. austroafricanus* (Barnard, 1940); *C. bacescui* Guţu, 1975; *C. caraspinosus* Dojiri and Sieg, 1997; *C. curticarpus* Băcescu, 1982; *C. kudinovae* Băcescu, 1981; *C. laubieri* Băcescu, 1982; *C. longissimus* Lang, 1968; *C. menziesi* Guţu, 1975; *C. oculicornutus* Lang, 1968; *C. prospectnes* Larsen, 1999; *C. romanae* Băcescu, 1987; *C. rotundirostris* Kudinova – Pasternak, 1989; *C. serratosipinosus* Lang, 1968; *C. simplicirostris* (Norman and Stebbing, 1886); *C. spinigena* Bamber, 2007; *C. varindex* Bamber, 2007.

Carpoapseudes heardi **n. sp.**

Material Examined

Holotype (USMN 1110765) 8.3mm male, Offshore Tobago, Block 22, 30 November 2006, Station SLCA-MDFD-1, 11°29'39.012" Latitude, 60°37'57.1494"Longitude, 537 m, mud.

Paratypes

4 specimens (11.2, 10.5 mm ovigerous females, 10 mm male, 6.4 mm juvenile) Offshore Tobago, Block 22, 30 November 2006, Station SLCA-MDFD-1, 11°29'39.012" Latitude, 60°37'57.1494" Longitude, 537 m, mud. USMN 1110766 2 specimens (10.3 mm male, 8.5 mm female) Offshore Tobago, Block 22, 16 November 2006, Station ASC-FRFD-6, 11°29'50.3394" Latitude, 60°47'56.3214" Longitude, 421 m, sandy mud. USMN 1110767

Diagnosis

Rostrum triangular, acuminate, extending beyond eye lobes, carapace without lateral apophyses. Eye lobes acuminate, anteriorly projecting. Epistome without spine. Labrum with two lobes on article 1. Male chela with ventral process on carpus. Pereonite 1 with bulbous expansion housing coxa. Pereonites 2 and 6 with small anterolateral spines Pereopods 2 and 3, basis with dorsoproximal spur-like apophysis. Pleopods inserted midlaterally, bearing exopods, endopods 1-articulate. Female with five pairs of oostegites located on cheliped and pereopods 1-4.

Etymology

This species is named for our friend and tanaid expert, Richard W. Heard.

Description of male holotype (Figures 1-3)

Body (Figure 1b) 8.3 mm long, glabrous, white, iridescent, tapering posteriorly, with long, narrow pleotelson. Carapace about 0.9 times as long as wide, with slightly downturned, triangular rostrum extending about 1/3 length of antennular article 1, inflated branchial lobes lacking spines, delineated dorsally by shallow furrows. Pereonite 1 widest, with large bulbose lateral lobes surrounding coxa of pereopod 1. Pereonites 3-5 with strong lateral spines located slightly anterior to the midpoint of the somite. Pereonites 2 and 6 with smaller anterolateral spines. Pleonites with epimera bearing spines and short setae. Hyposphenia present between cheliped bases, on all pereonites and on pleonites 1-5. Genital cone broadly rounded. Pleopods inserted midlaterally.

Antennule (Figure 1c) with 4-articulate peduncle, article 1 longer than remaining three combined, with several broom setae on outer margin, two adjacent setae distally. Inner margin with two long and several short setae. Article 2, outer margin with cluster of setae and aesthetascs on distal third. Article 3 shorter than article 2, with long seta on outer distal margin and shorter paired setae on inner distal margin. Article 4 common with outer flagellum of 20 articles, with strap-like aesthetascs on inner margins of articles 4, 6, 8, 9, 10, 11-18, on outer margins of articles 11, 13, and 17. Single setae on outer margins of articles 5, 7 and 9. Tip of outer flagellum with four simple setae and one broom seta. Inner flagellum of six articles with single simple and broom seta on third article, two simple setae on fifth article, strap-like aesthetasc and three simple setae on terminal article.

Antenna (Figure 1d) with 5-articulate peduncle. Article 1 with broad inner lobe, article 2 with squama and three setae on outer margin, article 3 shortest, with single inner distal seta, articles 4-5 with several broom setae. Squama bearing five setae, with one on inner margin and four on outer margin. Flagellum 9-articulate, with 5 terminal setae. Articles 1, 2 and 4 with long outer seta, article 7 with broom setae.

Labrum (Figure 1e) with article 1 bearing two ventral

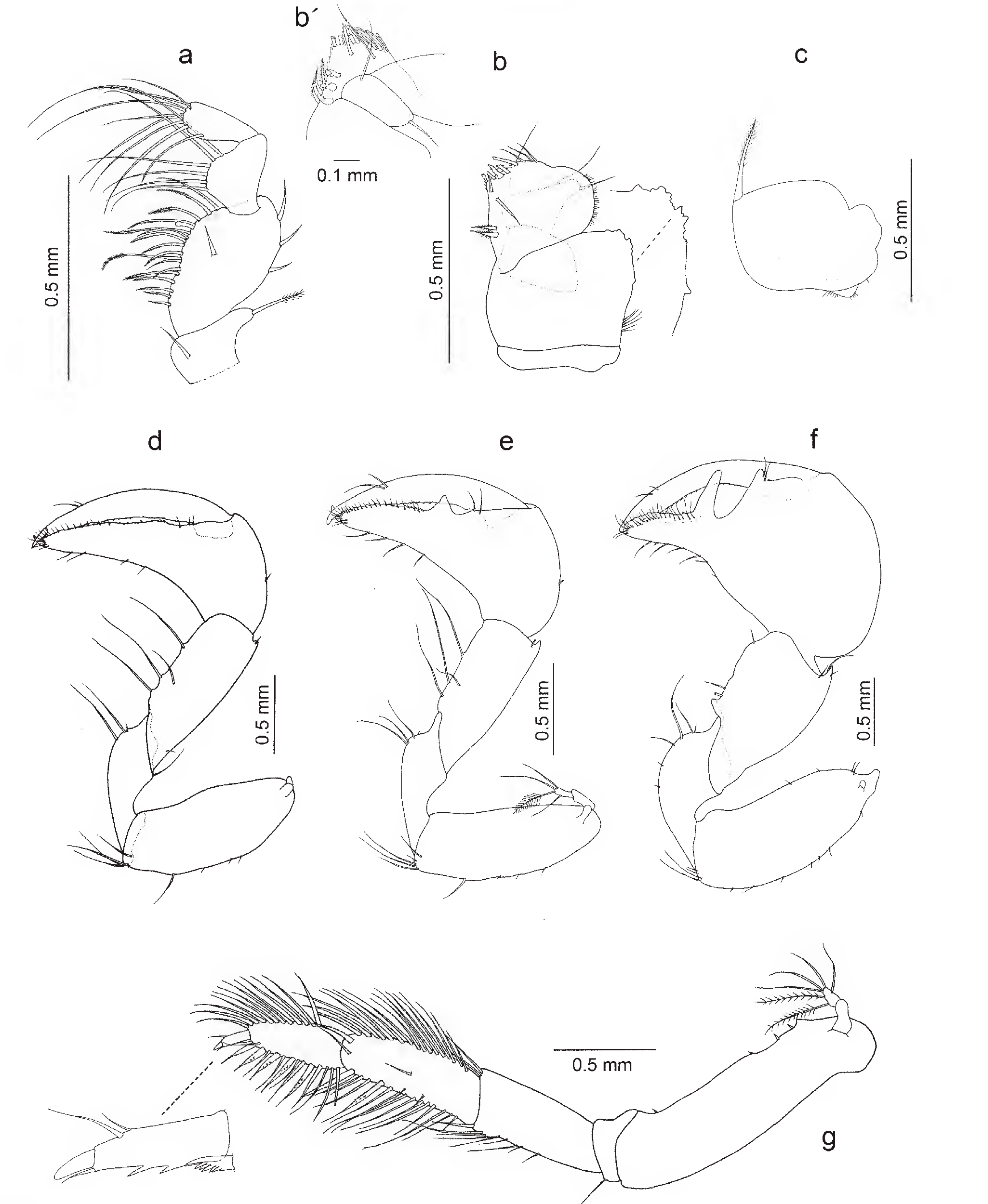

Figure 2.
Carpoapseudes heardi n. sp. (a) male holotype (8.3 mm) maxilliped palp, (b) male paratype(10 mm) maxilliped endite, (b´) endite opposite, (c) epignath, (d) ovigerous female paratype (11.2 mm) cheliped, (e) cheliped, (f) male paratype (10 mm) cheliped, (g) pereopod 1.

lobes, each with long setal cluster on outer margin, article 2 with broadly excavate mid-margin.

Mandible (Figures 1f-h). Left mandible with broad molar process bearing oval, smooth triturative surface with outer cusp and fine setules. *Pars incisiva* with five broad denticles. *Lacinia mobilis* narrow, with four denticles. Spine row con-

sisting of five setae, four with multifurcate tips, one with bifurcate tip. Palp 3-articulate, article 1 shortest, bearing two setae. Article 2 longer than articles 1 and 3 combined, distal inner margin bearing row of seven serrated setae, decreasing in length distally, with six smaller, curved, subequal setae. Article 3, distal two thirds bearing inner row of thirteen curved setae, distal margin bearing two longer setae. Right mandible not illustrated but similar to left.

Labium (Figure 1i). Basal endite not illustrated. Palp, margins with long setules, with two distal setae.

Maxillule (Figures 1j-k). Outer endite with nine distal spiniform setae, two subterminal setulate setae. Palp 2-articulate, distal article longer than proximal, bearing facial row of four elongate subterminal setae with serrated tips and single larger terminal spine (broken). Inner endite with five distal setae with fine setules on inner and outer margins

Maxilla (Figure 1l). Movable endite, outer lobe with seven setae, outermost pair setulate, inner lobe with five stout curved setae. Fixed endite, outer lobe with two subterminal setae, one palmate, three multifurcate setae, six setae with serrate tips. Inner lobe with comb row of about 25 curved, basally inflated setae, with one to two straight guard setae.

Maxilliped (Figures 2a-b,b´). Coxa very short, nearly as wide as basis. Basis large, quadrate, distal outer margin crenulate, sculptured. Palp article 1, outer lobe bearing large, distally setulate seta, inner margin with single simple seta. Article 2, outer margin bearing three stout curved setae, lateral surface with single seta, inner margin with two rows of setulate setae, dorsal row consisting of six short and single, atypical, thick, blunt seta, ventral row with eleven setae. Article 3 with six distally attenuate setae on inner margin. Article 4 with seven setae. Endite, inner margin serrate, with three coupling hooks, three circum-plumose setae. Inner surface with subterminal, apically inflated serrate spine ("leiopid spine") and subterminal simple setae. Distal inner margin with five truncate setae, outermost seta distally setulate. Distal outer margin of endite with five setae, distally flattened.

Epignath (Figure 2c). Cup-shaped, with large setulate seta and two basal lobes.

Cheliped (Figures 2d-f). Basis short, bearing 2-articlulate exopod, article 2 with four plumose setae. Ventral margin of basis without spine, with five ventrodistal setae. Merus with three ventrodistal setae and short spine. Carpus, ventral margin with three long and one short seta, dorsodistal margin with spinous apophysis and single strong seta (broken). Propodus, fixed finger with ventral margin bearing four short setae, distal margin with small setae and claw, setal row extending onto cutting surface, triangular tooth on cutting margin followed by row of short spines. Dactylus, cutting margin bearing short spines, dorsal margin with two setae.

Pereopod 1 (Figure 2g). Coxal process bulbous, as long as pereonite 1, with anterolaterally directed spine. Basis 3.7 times longer than wide, with 2-articulate exopod, article 2 with five plumose setae (illustrated for 2 setae). Dorsoproximal margin of basis with apophysis and short setae. Ischium with ventral seta. Merus about 4/5 length of carpus, ventrodistal margin with eight setae, one stronger spine, dorsodistal margin with two setae. Carpus, both margins lined with row of long attenuate setae, ventrodistal margin with two spinulate spines. Propodus about 2/3 length of carpus, dorsal margin with eleven attenuate setae, ventral margin with six antennuate setae, seven spinulate spines, and single short distal spine bearing seven ventral denticles. Dactylus plus unguis short, 1/3 length of propodus. Dactylus, ventral margin with two spines, one distal seta, dorsodistal margin with two setae. Unguis thick, shorter than dactylus.

Pereopod 2 (Figure 3a). Basis with spur on dorsoproximal margin, numerous short setae on both margins. Ischium with ventrodistal seta (broken). Merus shorter than carpus, with six ventral and three dorsal setae. Carpus with ventral margin with nine long attenuate setae and dorsodistal margin with distal cluster of four setae. Lateral surface of carpus with six setae with raised bases. Propodus shorter than carpus, with ventral row of six setae and short distal spine. Propodal spine with six teeth on ventral margin and with fine dorsal denticles. Distal half of dorsal margin of propodus with closely spaced row of eight long, attenuate setae. Dactylus about ½ length of propodus, with single dorsal and ventral setae. Unguis broken.

Pereopod 3 (Figure 3b). Basis with dorsoproximal spur, longer than combined length of ischium, merus and carpus. Ischium short, with dorsodistal broom seta. Merus shorter than carpus, with few setae. Carpus similar to pereopod 2, but without row of lateral setae, ventral margin with six setae, one ventrodistal spine. Propodus with setae and broom seta on dorsodistal half. Ventral margin of propodus with five setae and short distal spine spine with three ventral teeth. Unguis plus dactylus slightly longer than propodus. Dactylus with one ventral fixed spine and three dorsal setae. Unguis slightly shorter than dactylus, weakly curved.

Pereopod 4 (Figure 3c). Basis like pereopod 3, but without spur and with constriction near middle (artifact?) with strong ventrodistal seta (broken). Ischium with two ventrodistal setae. Merus about ½ length of carpus with single dorsodistal and three ventrodistal setae. Carpus longer than all other articles except basis, with lateral oblique row of seven setae on distal margin and with four setae on ventral margin. Propodus with dorsoproximal broom seta, distal margin truncate, bearing five attenuated setae. Ventral margin with three setae. Dactylus plus unguis shorter than that of pereopod 3, about 2/3 length of propodus. Dactylus longer than unguis, dorsal margin with two setae, ventral margin with one distal seta.

Pereopod 5 (Figure 3d). Basis bearing large mid-dorsal broom seta, two ventrodistal setae. Ischium similar to pereopod 2. Merus shorter than carpus, with single ventrodistal and dorsodistal setae. Carpus with single strong ventrodistal

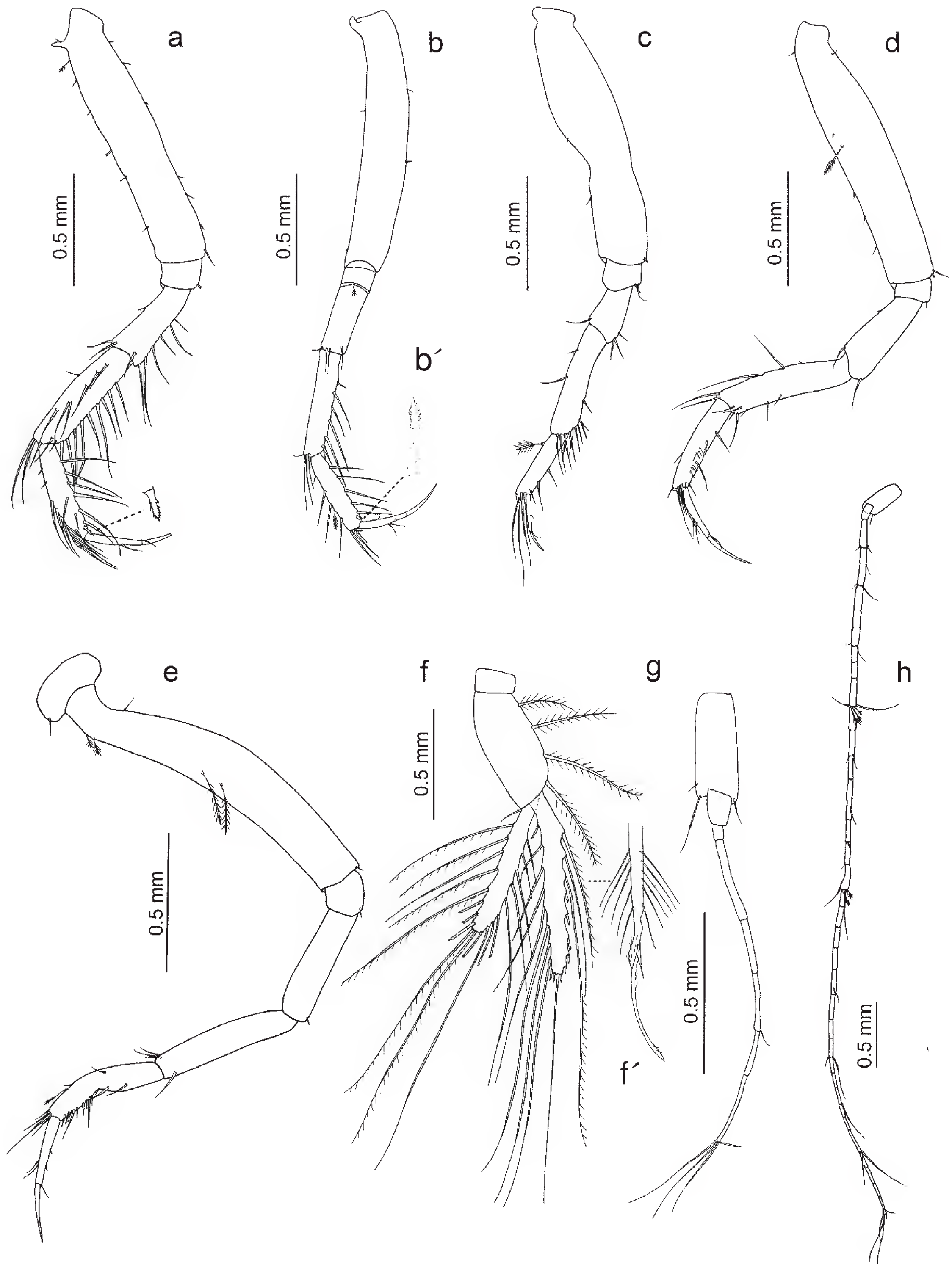

Figure 3.
Carpoapseudes heardi n. sp. (a) male holotype (8.3 mm) pereopod 2 with enlarged spine, (b) pereopod 3 with enlarged spine, (b´) ovigerous female paratype (11.2 mm) pereopod 3 spine, (c) pereopod 4, (d) pereopod 5, (e) pereopod 6, (f) pleopod 1, (f´) male paratype (10 mm) pleopod endopod specialized seta, (g) female paratype uropodal exopod, and (h) male paratype endopod.

seta and four weaker setae. Dorsal margin of carpus with four attenuated setae on distal half. Mesiodistal margin of carpus with two setae. Propodus about equal to length of merus, with ventrolateral row of six spines on inner margin. Mesiodistal margin of propodus with four setae. Ventral margin of propodus with four setae. Dactylus plus unguis equal to length of propodus. Dactylus, with two dorsal and single ventral setae. Unguis slightly shorter than dactylus.

Pereopod 6 (Figure 3e). Coxa toroidal-shaped, distal margin bearing single seta. Basis with two large middorsal broom setae with smaller and distal broom setae present. Ischium with single ventrodistal seta. Merus with ventrodistal seta. Carpus longer than merus, with few setae. Propodus with midventral indention (artifact?), 10 spines, two setae. Dorsal margin with broom seta on distal third, with five distal setae. Dactylus plus unguis longer than propodus. Ventral margin of dactylus with four setae and dorsal margin with two setae. Unguis shorter than dactylus.

Pleopod (Figures 3f,f´). Peduncle 2-articled, article 1 short, naked, rectangular, article 2 with four plumose setae on inner margin. Endopod slightly longer than exopod, 1-articled, margins lined with 20 or more long plumose setae. First proximal seta of inner margin of endopod highly modified (Figure 3f). Exopod 1-articulate, margins bearing about 20 plumose setae.

Uropod (Figures 3g-h). Protopod attached near dorsal margin of pleotelson, dorsal margin with two distal setae, ventral margin with single distal seta. Exopod with seven articles, including three pseudosegments, bearing one seta on article 5. Tip with four setae. Endopod with 33 articles, including about three pseudosegments. Articles 2-4, 7-12, 16, 19, 27, and 26 with setae, usually paired. Articles 7 and 13 with 2-3 broom setae. Tip with five setae.

Sexual and developmental differences

Females are similar to males in dorsal view (Figure 1a). Cheliped merus of female has two ventrodistal setae (Figure 2d). Large males have a tubercle on the ventroproximal margin of the carpus of the cheliped that is absent in younger males and all females (Figure 2f). Propodus of largest males has more setae on the ventral margin of the fixed finger. The large propodal tooth on the cheliped fixed finger is absent in the female. The female cheliped of the *C. heardi* n. sp. lacks the propodal tooth although this is prominent in the younger and older males. In the largest adult males, a second smaller cheliped tooth is also found near the articulation with the dactylus (Figure 2f). Large males have proportionally short exopods on the chela when compared to the length of the basis. In the ovigerous female pereopod 3 has a short propodal spine bearing smaller denticles. Hyposphenial spines are stronger in the large adults. Numerous aesthetascs, strap-like in shape, are also found on the male outer antennular flagellum, although these are absent in the female. With regards to the base of the maxilliped, the outer margin of article 2 of mature *C. heardi* n. sp. has strong crenulations as also illustrated for *C. auritochelis* Kudinova–Pasternak, 1975. These and the size of the cheliped propodal triangular tooth are likely related to the maturity of the tanaid.

Ovigerous females with a marsupium have five pairs of oostegites present including one on the cheliped and one on pereopods 1-4. Large males also have five plumose setae on the inner margin of the protopodite of the pleopod rather than four. The highly modified seta on the endopod of the pleopod also seems to be found only in mature individuals (Figure 3f´).

Discussion

Carpoapseudes heardi n. sp. differs from all other known species in the genus by a combination of characters, the prominent being presence of parallel eyespines and the lack of an epistomal spine. *Carpoapseudes bacescui* Guţu, 1975 and *C. simplicirostris* (Norman and Stebbing, 1886) also have parallel or anteriorly directed eyespines plus epistomal spines. *Carpoapseudes heardi* n. sp. further differs from these two species by the presence of a larger rostrum. In *C. simplicirostris* the rostrum is very narrowed and stick-like while the rostrum of *C. bacescui* is shorter than that of *C. heardi* n. sp. Like *C. heardi* n. sp., *C. longissimus* Lang, 1968 and *C. oculicornutus* Lang, 1968 lack epistomal spines, but they differ from the new species by having divergent rather than parallel eyespines. A comparison of the first pereopod of *C. heardi* n. sp. to other related species reveals that this leg has a shortened dactylus similar to that of *C. serratospinosus* Lang, 1968, but the merus of *C. heardi* n. sp. lacks the dorsal setae as characteristic of the former species.

Carpoapseudes prospectnes Larsen, 1999 which was described from Brazilian waters and is the only other member of the genus known from the western Atlantic, is distinguished from *C. heardi* n. sp by the having an epistomal spine and lacking parallel eyespines. With the description of *C. heardi* n. sp. the genus *Carpoapseudes* now contains 18 species, most of which are reported from depths greater than 1000 m. Three species of *Carpoapseudes*, *C. heardi*, n. sp., *C. prospectnes*, and *C. serratispinosus* however, are known from depths < 900 m, with *C. prospectnes* having the shallowest occurrence (295-360 m).

An examination of the mouthparts of the new species reveals that the inner distal seta on the maxilliped endite of *C. heardi* n. sp. is long, serrated, and not leaf-like, but can be considered a "leiopid spine" based on its size and position, indicating a relationship with *Leviapseudes* Sieg, 1983 and related genera. The labial palp of *C. heardi* n. sp. has two terminal spines as in *C. spinigena* Bamber, 2007 and *C. prospectnes* whereas three spines are present in most other species. The mandibular palp of the new species has an interesting arrangement of setae on the second article of the palp, with the setae steadily decreasing in size distally on the article.

The shape of the pereopod 3 propodal spine of *C. heardi* n. sp., with regards to the size and number of ventral denticles, also varies between the sexes. Females have smaller and more numerous ventral denticles than males (Figure 3b´).

The number of oostegites present in members of *Carpoapseudes* is controversial. Guţu (1981) and Dojiri and Sieg (1997) reported that the genus *Carpoapseudes* had five pairs of oostegites; however, Larsen (1999) reported only 4 pairs. It is possible that some authors did not have fully ovigerous female material available to study. Based on our observations, the cheliped oostegite of *C. heardi* n.

sp. does not form until the marsupium is present. For instance, one ovigerous female (11.2 mm; Figure 1a) did not have a cheliped oostegite whereas another (10.5 mm) had a fully formed marsupium with the cheliped oostegite. Finally, on C. *heardi* n. sp. the oostegite forms posterior and medial to the insertion of the cheliped and not near the exopod as in other apseudid genera examined by the authors.

Like most species of the genus, C. *heardi* n. sp. has simple non-plumose setae of the lateral epimera of the pleonites; in contrast, C. *bacescui* Guţu, 1975 has long plumose setae.

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Literature Cited

Băcescu, M. 1982. *Carpoapseudes laubieri* sp. n. et C. *curticarpus* sp. n. de l´Atlantique de NE (Bassin ouest–européen) et quelques détails nouveaux sur La calabilité Du genre. Travaux du Museum national d'Histoire naturelle "Grigore Antipa" 24:55–68.

Bamber, R N. 2007. Suborders Apseudomorpha Sieg, 1980 and Neotanaidomorpha Sieg, 1980 In: K. Larsen and M. Shimomura, eds. Tanaidacea (Crustacea: Peracarida) from Japan III. The deep trenches; the Kurile–Kamchatka Trench and Japan Trench. Zootaxa Monograph # 1599, Magnolia Press, Auckland, New Zealand, p. 13-40.

Beddard, F.E. 1886. Report on the Isopoda collected by H.M.S. Challenger during the years 1873-1876. Part II. The Voyage of H.M.S. Challenger. Zoology 17: 1-175, Pls I-XXV.

Dojiri, M. and J. Sieg. 1997. The Tanaidacea. In: J.A. Blake and P.H. Scott, eds. Taxonomic Atlas of the Benthic Fauna of the Santa Maria Basin and Western Santa Barbara Channel. Volume 11. The Crustacea Part 2. The Isopoda, Cumacea and Tanaidacea. Santa Barbara Museum of Natural History. Santa Barbara, CA, USA, p. 181-278.

Guţu, M. 1972. Phylogenetic and systematic considerations upon the Monokonophora (Crustacea–Tanaidacea) with the suggestions of a new family and several new subfamilies. Revue Roumaine de Biologie (Serie Zoologie) 17:297–305.

Guţu, M. 1975. *Carpoapseudes bacescui* n. sp. and C. *menziesi* n. sp. from the Peru-Chile trench. Revue roumain de Biologie (Biologie animale) 20:93– 100.

Guţu, M. 1980. On the status of the "groups" Leiopus and Carpoapseudes (Crustacea, Tanaidacea) and their systematic position. Travaux du Museum national d'Histoire naturelle "Grigore Antipa" 22:385-392.

Guţu, M. 1981 A new contribution to the systematics and phylogeny of the suborder Monokonophora (Crustacea, Tanaidacea). Travaux du Museum national d'Histoire naturelle "Grigore Antipa" 23:81-108.

Guţu, M. 1996. The description of ***Spinosapseudes*** n. g., and amended diagnoses of two genera of Tanaidacea (Crustacea). Revue Roumaine de Biologie (Série de Biologie Animale) 41:87–93.

Kudivova-Pasternak, R.K. 1975. Tanaidacea (Malacostraca) of the deep-sea Romansh and Guinea Hollow. Zoologicheskii Zhurnal 54:682-687.

Lang, K. 1968. Deep–sea Tanaidacea. Galathea Report 9:23–209.

Larsen, K. 1999. A new species of the deep-sea genus *Carpoapseudes* Lang from the southwestern Atlantic (Crustacea, Tanaidacea) Zoostema 21:647–659.

Larsen, K. 2003. Proposed new standardized anatomical terminology for the Tanaidacea (Peracarida). Journal of Crustacean Biology 23:644–661.

Larsen, K. 2005. Deep sea Tanaidacea (Peracarida) from the Gulf of Mexico. Crustaceana Monographs 5, Brill, Boston, MA, USA, 381 p.

Norman, A.M and T.R.R. Stebbing. 1886. On the Crustacea Isopoda of the 'Lightning, 'Porcupine' and 'Valorous' Expeditions. Transactions of the Zoological Society of London 12 (Part IV, No. 1): 7-141, Pls 16-27.

Sieg, J. 1983. Tanaidacea. In H.-E. Gruner and L.B. Holthuis, eds. Crustaceorum Catalogus, Vol 6, Dr. W. Junk Publishers Inc., Hague, The Netherlands, 552 p.

Watling, L. 1989. A classification system for crustacean setae based on the homology concept. In: B.E. Felgenhauer, L. Watling, and A.B. Thistle, eds. Functional Morphology of Feeding and Grooming. Crustacean Issues 6, Balkema, Rotterdam, The Netherlands, p. 15–27.

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SHORT COMMUNICATION

AN UNUSUAL REACTION AND OTHER OBSERVATIONS OF SPERM WHALES NEAR FIXED-WING AIRCRAFT

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Introduction

Data on the reactions by cetaceans to aircraft flying overhead (or in the near vicinity) are limited (e.g., Richardson et al. 1995, Patenaude et al. 2002). This information is important for assessing potential effects of aircraft on federally protected species, such as sperm whales (*Physeter macrocephalus*) particularly in association with offshore oil and gas exploration in the northern Gulf of Mexico (NGOM) and elsewhere. As noted in the draft recovery plan for the sperm whale, "the severity of the threat is unknown for sound-producing factors (including aircraft) related to the oil and gas industry" (NMFS 2006). Sperm whales in the NGOM are well-known to occur in areas of intense oil and gas exploration and development activities (e.g., Jochens et al. 2006). Helicopters (as well as work boats) are used to transport workers to and from operating offshore platforms in the NGOM. These helicopter operations occur between the water's surface and altitudes of ~2,135 m (e.g., Daskalakis and Martone 2004). Low altitudes are flown during approaches to and departures from offshore platforms. The NOAA Fisheries currently includes in its biological opinions, a conservation recommendation that permit holders maintain helicopter traffic over the NGOM at altitudes above 305 m, if practicable, to avoid disturbance to whales and sea turtles. It is projected that an average rate of 25,000-55,000 helicopter operations will occur annually in the Central Planning Area (including the Mississippi River Delta area, a known high-use area by sperm whales, particularly females and their calves) (MMS 2006). The frequency of such flights is anticipated to continue increasing as the number of operating offshore structures increase.

Reported behavioral reactions by sperm whales to aircraft are sparse, highly variable, and largely anecdotal as summarized in Table 1. Observers since the whaling era began have noted that sperm whales tend to be skittish (Whitehead 2003). When documented, sperm whale reactions to both planes and helicopters range from no reaction (Clarke 1956, Gambell 1968, Green et al. 1992) to reactions such as increased surface intervals and dramatic behavioral changes (Clarke 1956, Fritts et al. 1983, Mullin et al. 1991, Würsig et al. 1998, Richter et al. 2003, 2006). Given the lack of supporting data for either case, it is important that these types of data are collected and consolidated into a cohesive document. Therefore, the specific objectives of our paper are to report our visual observations of sperm whale reactions to straight-line aircraft fly-bys (i.e., passes), to report a unique observation of a recognized "stress behavioral reaction" exhibited by sperm whales during an overhead circling by small fixed-wing aircraft, and to provide a summary review of published related studies.

Materials and Methods

Cetacean observations were made during a series of multi-year, line-transect aerial surveys for cetaceans conducted within 45 km from shore of the main Hawaiian Islands. Specifics of the survey protocol and general area descriptions are detailed in Mobley et al. (2000). Briefly, surveys occurred over waters less than 2,000 m in bottom depth (Mobley et al. 2000) using small aircraft (1993, Cessna 172; 1994 and 1995, Skymaster; and 1998, Partenavia) at an altitude of 245 m and a speed of 185 km/hr. Four personnel were aboard the aircraft during all flights: a pilot, a data recorder, and two observers. Time, location and altitude information were recorded in real time using a computer linked to an altimeter and global positioning system every 30 sec and manually whenever a sighting occurred. A Hi 8-mm video camera and a 35-mm camera with 300 mm telephoto lens were used to document unusual sightings and behavior.

Response (reaction or no reaction) by cetaceans during an initial pass was noted as required by the NMFS research permit obtained for the surveys. A "reaction" to the aircraft was defined as an overt change in the initially observed orientation or behavior of at least one animal in a group; for example, an abrupt dive associated with a splash or display of the tail flukes, a breach, a tail slap, etc. (similarly described by Green et al. 1992, Würsig et al. 1998, Patenaude et al. 2002). After the initial pass of a sighting, the aircraft sometimes circled over or near (usually within ~200 m lateral distance) the sighting so that observers could verify species, group size, and species composition. For the purposes of this report, "adult" refers to non-calf whales.

TABLE 1. *Summary of available information on reactions of sperm whales to low altitude overflying aircraft.*

Location	Aircraft	Altitude (m)	Behavior Description	Reference
the Azores	Sikorski S55 helicopter	~4.6-27 m	4 sperm whales marked with 'Discovery-type' marks shot from a standard rifle during two different flights; apparent "unconcern" until a down-draught of rotors caused much agitation of the water, causing the whales to quickly dive and simultaneously defecate	Clarke (1956)
South Africa	Cessna 310H	~150 m	Whales seemed unaware of aircraft flying overhead (number not specified)[a]	Gambell (1968)
northern Gulf of Mexico	Beechcraft (Model AT-11)	alternating altitudes of 91 m and 229 m	Circling often (number not specified) disturbed whales by causing changes in direction, dive patterns, and increased speed of movement; one observation of breaching possible response	Fritts et al. (1983)
northern Gulf of Mexico	Twin Otter	~230 m	Some (number unspecified) whales affected by flyovers and dove immediately while other animals remained at the surface	Mullin et al. (1991)
Oregon and Washington	DeHavilland Twin Otter	~65 m	No reaction by 24 observed groups	Green et al. (1992)
northern Gulf of Mexico	Twin Otter	Usually ~230 m	7 (28%) of 25 groups changed behavior when approached to within 305 m	Würsig et al. (1998)
the Bahamas	Cessna 172	50-245 m	Group of six sperm whales (including one calf) closed ranks and one whale turned on its side to apparently look up towards aircraft circling overhead	C. MacLeod, pers. comm., Beaked Whale Research Project, University of Aberdeen, Lower Right, 59 Jute Street, Aberdeen, AB24 3EX, U.K.
Kaikoura, New Zealand	Fixed-wing aircraft	Aircraft recorded as present when flying circular pattern at least 150 m above whale(s)	Study of impacts of aircraft-based whale-watching on male sperm whales from small vessel (n=116) and from shore (n=29). Transient males delayed time to first click (vocalization) and reduced surfacing time near aircraft, while residents slightly increased their surface time near aircraft. No alteration of frequency of heading changes by residents or transients. Results indicated aircraft presence combined with other factors (e.g., season, year) contributed to slight changes in behavior.	Richter et al. (2003, 2006)
Kauai, Hawaii	Cessna 172, Skymaster, Partenavia	~233-269 m	3 of 8 groups (<360 m lateral distance) reacted to fly-by by abruptly diving. One group of 11 (including one calf) closed flanks, slowed down, formed a reverse marguerite with calf in middle, then dove while aircraft circled overhead for 6 min.	Present study

[a] *General statement based on review of daily diaries kept by pilots operating spotting aircraft associated with whaling operations during 1966, 1967 and 1968.*

Results

Data were obtained from observations of 24 sperm whale groups totaling 109 individuals (mean = 4.6 whales, sd = 5.3, range 1-20). An additional three sightings with no lateral distance data were excluded from analyses; none of these groups demonstrated a visible reaction to the aircraft. Nine calves were sighted in six of the 24 groups. Most (n = 13) of the 24 sightings were made from the Skymaster, 10 from the Partenavia, and 1 from the Cessna. During initial passes, aircraft altitude ranged from 233-269 m and lateral distance to whale sightings ranged from 103-3,427 m (n = 24).

Responses to aircraft passes

A reaction to the initial pass of the aircraft was observed during three (12%) of 24 sightings: two from the Skymaster (both single adult whales) and one from the Cessna 172 (a group of four adult whales). All three reactions consisted of a hasty dive and occurred < 360 m lateral distance from the aircraft. Of the eight groups seen < 360 m lateral dis-

tance from the aircraft, three (38%) reacted to the passing aircraft; no reactions were noted for the remaining 16 sightings at lateral distances > 360 m from the aircraft (n = 21). No reaction was observed during the two closest (103 m and 208 m lateral distance) initial passes (both by the Skymaster) (Figure 1A). However, a reaction by the closest of these initial sightings (103 m lateral distance) occurred during a subsequent resighting 3 min later while the Skymaster circled overhead. This response is described below and is based primarily on Hi 8-mm video, photographs, and field notes.

Response to circling aircraft

While surveying at 235 m altitude (50 km north of Kaua'i), a single sperm whale was sighted from the Skymaster and no reaction to the initial pass was seen. Subsequently, the aircraft turned to estimate group size and confirm species identification. During this time, the aircraft increased altitude and began circling the location of this individual to look for more animals. About 3 min later, a group of 11 sperm whales (10 adults plus 1 calf) surfaced in the same area. The aircraft continued circling this group for ~ 6 min at distances of 0-500 m (laterally) and altitudes of 245-335 m. All whales were visible at or near the water's surface throughout most of the observations. One adult estimated to be ~ 1/3 longer than the other adults and not associated closely with the calf was assumed to be a mature male (bull) based on its relative body length (Rice 1989). After the aircraft circled overhead for about 4 min, the whales ceased forward movement, moved closer together in a parallel flank-to-flank formation (Figure 1B), and formed a fan-shaped semi-circle with heads facing out and flukes toward the middle of the semi-circle (Figure 1C). The bull was on the left outer edge of the semi-circle and the calf remained near the middle of the group. Maximum distance between individuals over the course of the observation decreased from about six body lengths to one, thereby, concentrating the group as a whole around the calf. During this time, one whale was seen on its side with its mouth agape. The entire episode lasted about 9 min from initial sighting to the unique behavioral observation.

Discussion

We interpreted the aforementioned group's formations as an agitation, distress, and/or defense reaction to our circling aircraft. This interpretation is based upon behavioral events displayed by sperm whales in situations of distress, reacting to perceived or actual threats, such as killer whales (*Orcinus orca*) (e.g., Arnbom et al. 1987, Pitman et al. 2001), false killer whales (*Pseudorca crassidens*) (Palacios and Mate 1996), short-finned pilot whales (*Globicephala macrorhynchus*) (Weller et al. 1996, Pitman et al. 2001), sharks (Best et al. 1984), whalers (Nishiwaki 1962, Caldwell et al. 1966, Berzin 1971), and vessel approaches (Palacios and Mate 1996). The characteristic responses to killer whales are individuals coming to the surface, swimming fast toward one another, and clustering actively and tightly (Whitehead 2003), similar to the behavior we observed. The semi-circle "fan" formation we describe is similar to defensive "marguerite"- and "spindle"-like formations reported by other researchers (Nishiwaki 1962, Berzin 1971, Arnbom et al. 1987, Weller et al. 1996, Pitman et al. 2001). Weller et al. (1996) observed open-mouth behavior (akin to our observation) by sperm whales, and interpreted this as a discrete distress response to harassment by short-finned pilot whales, based on obvious distress behavior reported by other researchers. In our observations, the mouth agape may have been a distress response to our aircraft. This same whale was swimming on its side, possibly to look up at the aircraft.

The tight parallel formation we observed is often a precursor to socializing events (during which animals huddle together and rub against each other), but also to defensive responses such as the fan formation we observed (D.M. Palacios, NMFS/Pacific Fisheries, Environmental Laboratory, Pacific Grove, California, pers. comm.). Thus, huddling may provide an opportunity for information transfer and reassurance between group members. For the group we observed, this behavior might have increased defensive capabilities by minimizing exposure of the flanks (particularly the calf) to a perceived threat. Similar behaviors by a group of six sperm whales (including one calf) in the Bahamas occurred when a Cessna 172 passed, then circled directly over the group at an altitude of about 50-245 m (C.D. MacLeod, Beaked Whale Research Project, Lower Right, Aberdeen, AB24 3EX, United Kingdom, pers. comm.). The group closed ranks and one individual turned on its side to apparently look up towards the aircraft.

In general, it is difficult to identify behavioral reactions during brief observation periods such as short overflights by aircraft; furthermore, some subtle changes in behavior (i.e., in respiration) are not evident without statistical analysis (e.g., Richardson et al. 1995). Thus, it is possible that sperm whales we observed may have exhibited reactions we did not recognize or see because they occurred after we had passed.

Reactions of sperm whales to perceived threat stimuli may be context dependent. Berzin (1971) described three separate fright reactions related to the level of the perceived threat: dive, aggregate at surface, and flight/flee. Pitman et al. (2001) further suggested that sperm whales often dive in the presence of boats (perhaps a mild response) vs. aggregate if the threat is immediate, forming a rosette when groups are small (typically < 9 whales). The three apparent dive responses we reported may have been a "mild fright" response to the brief passes by our aircraft. In contrast, the two group formations we described appear to have been fright responses to persistent overhead circling by the aircraft and resemble the "spindle" group formed in response to an immediate perceived threat (Pitman et al. 2001).

Received sound levels of our aircraft near sperm whale

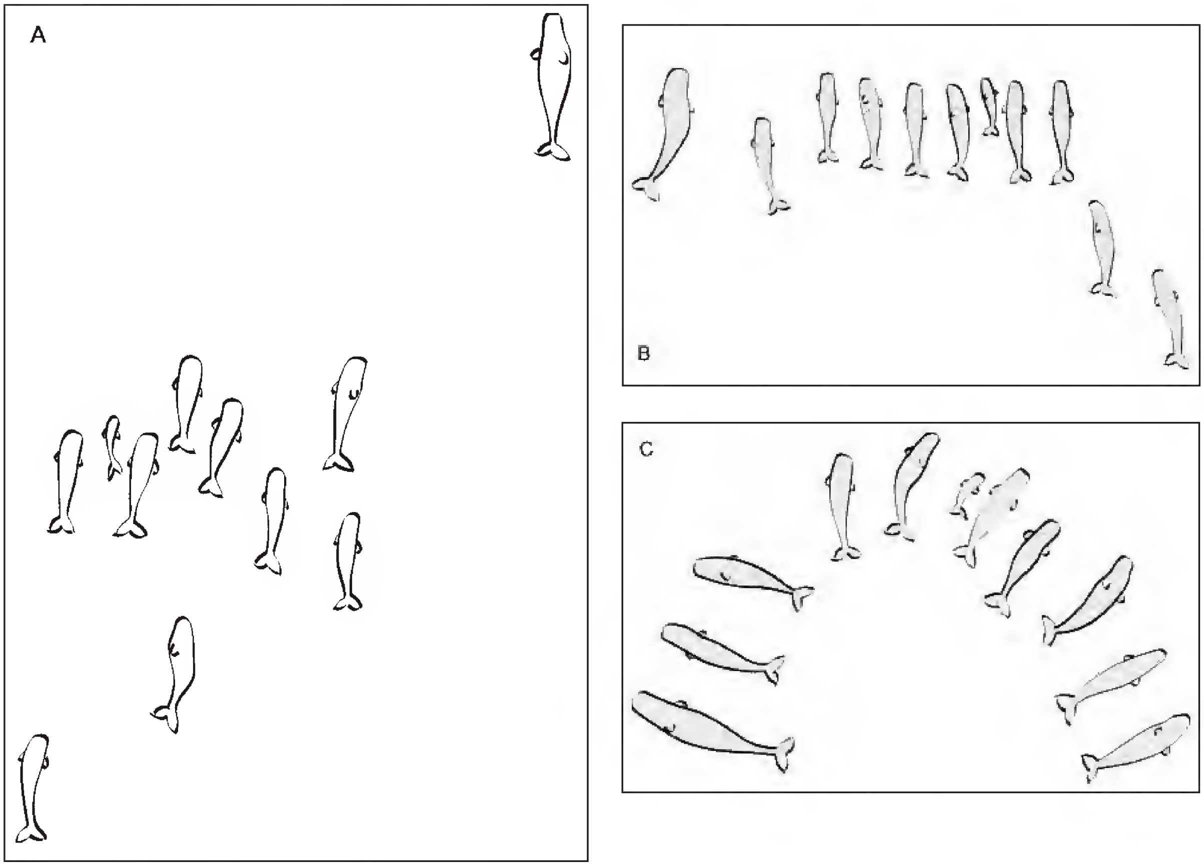

Figure 1.
Chronological group formations exhibited by a group of 11 sperm whales (including one bull and one calf) while a Skymaster aircraft circled overhead on 9 April 1994 from 1146h to 1155h (see text):
(A) No reaction, 1151h;
(B) Flank to flank parallel formation, 1153h;
(C) Semicircle formation, 1154h. Scale is approximate.

sightings were not available, and cannot be realistically calculated for our data, given the variation and complexities involved in estimating aircraft-to-surface and sub-surface sound propagation (see Richardson et al. 1995). However, available data indicate that the expected frequency range and dominant tones of sound produced by our aircrafts overlap with the known low-end frequency range of sperm whale vocalizations (< 0.1 to 30 kHz; see summaries by Richardson et al. 1995 and Ketten 1998). Snell's Law predicts a 26° sound cone from the vertical for the transmission of sound from air to smooth-surface water (Urick 1972, Richardson et al. 1995). The angle of this cone becomes greater in Beaufort wind force > 2. Based on altitudes, the group of 11 sperm whales with the unusual reaction described above presumably received both acoustic and visual cues (the aircraft and/or its shadow) from the circling aircraft, as they were located directly under the aircraft and/or well within Snell's predicted sound cone. The other 24 sperm whale groups that were passed once by our aircraft were outside (104–3,427 m lateral distance) the theoretical 26° sound cone (lateral distance 54–62 m); however, whales near this sound cone (within roughly several hundred meters) may have heard the overflying aircraft via scattering associated with the rough sea surface at the time (Beaufort wind force 3-4).

Based on other studies of cetacean responses to sound (Richardson et al. 1995, Patenaude et al. 2002), we believe that our observed reactions to brief overflights by the aircraft were short-term and probably of no long-term biological significance. Although isolated occurrences of this type are probably not biologically significant, repeated or prolonged exposures to aircraft overflights have the potential to result in significant disturbance of biological functions, especially in important nursery, breeding or feeding areas (Richardson et al. 1995). Activities involving aircraft that might result in harassment of sperm whales include military training exercises, helicopter overflights associated with offshore oil and gas exploration and development (for example, in the NGOM), recreational/ecotourism flights (for example, off Hawaii and New Zealand) and research surveys.

This limited description sheds light on the need to systematically document behavioral responses by cetaceans to aircraft, particularly by protected species, such as the endangered sperm whale. There is also a need to document received sound levels of aircraft by whales, and to record and

compare whale behavior before, during and after controlled overflights, ideally of the same individual(s), to provide increased statistical power to account for the inherent variation among individuals. The latter approach has been used during land-based observations of humpback whales circled by research aircraft near Hawaii (Smultea et al. 1995) and to some extent from land-based sites and small vessels where sperm whales occur near shore (Richter et al. 2003). It is typically difficult to determine the reactions of cetaceans to overflights, since most observations have been from the disturbing aircraft itself (Richardson and Würsig 1997) or a small nearby vessel. These observation platforms limit and potentially confound what can be observed, and can preclude isolated comparison of behavior before, during, and after aircraft disturbance. Such data could also be collected by tracking whales with non-invasive tags (such as the D-tag developed by Johnson and Tyack 2003) capable of recording received sound levels and water depth among other data (such as changes in orientation of the animal in the water); this technique could ideally be combined with non-intrusive behavioral observations (e.g., theodolite tracking from shore).

In summary, based on our and others' observations, the biological significance or consequences of the potential impact of aircraft overflights on cetaceans warrants further, ideally systematic studies. These studies should be conducted with the following goals: consideration with respect to environmental planning purposes; implementation of monitoring and mitigation measures; and deliberation in decision-making regarding regulations affecting marine mammals.

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Literature Cited

Arnbom, T., V. Papastavrou, L.S. Weilgart, and H. Whitehead. 1987. Sperm whales react to an attack by killer whales. Journal of Mammalogy 68(1):450-453.

Berzin, A.A. 1971. Kashalot. Izdatel'stvo pishchevaya promyshlennost, Moskova (The sperm whale). (English translation, 394 pp). Israel Program for Scientific Translations, Jerusalem.

Best, P.B., P.A.S. Canham, and N. Macleod. 1984. Patterns of reproduction in sperm whales, *Physeter macrocephalus*. Reports of the International Whaling Commission, Special Issue 6:51-79.

Caldwell, D.K., M.C. Caldwell, and D.W. Rice. 1966. Behavior of the sperm whale, *Physeter catodon* L. In: K.S. Norris, ed. Whales, Dolphins, and Porpoises. University of California Press, Berkeley, CA, USA, p. 677-717.

Clarke, R. 1956. Marking whales from a helicopter. Norsk Hvalfangst-Tidende 45:311-318.

Daskalakis, A. and P. Martone. 2004. Deployment and Evaluation of the Helicopter In-Flight Tracking System. Technical Report. NASA/TM–2004-213447. National Aeronautics and Space Administration, Moffett Field, CA, USA.

Fritts, T.H., A.B. Irvine, R.D. Jennings, L.A. Collum, W. Hoffman, and M.A. McGehee. 1983. Turtles, birds, and mammals in the northern Gulf of Mexico and nearby Atlantic waters. FWS/OBS-82/65. Technical Report. U.S. Fish and Wildlife Service, Washington, D.C., USA.

Gambell, R. 1968. Aerial observations of sperm whale behaviour. Norsk Hvalangst-Tidende 57:126-138.

Green, G.A., J.J. Brueggeman, R.A. Grotefendt, C.E. Bowlby, M.L. Bonnell, and K.C. Balcomb III. 1992. Cetacean distribution and abundance off Oregon and Washington, 1989-1990. In: J.J. Brueggeman, ed. Oregon and Washington Marine Mammal and Seabird Surveys. OCS Study MMS 91-0093. Minerals Management Service, Pacific OCS Region, Los Angeles, CA, USA, p. 1-100.

Jochens, A., D. Biggs, D. Engelhaupt, J. Gordon, N. Jaquet, M. Johnson, R. Leben, B. Mate, P. Miller, J. Ortega-Ortiz, A. Thode, P. Tyack, J. Wormuth, and B. Würsig. 2006. Sperm whale seismic study in the Gulf of Mexico, Summary Report, 2002-2004. OCS Study MMS 2006-034. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA, USA.

Ketten, D.R. 1998. Marine mammal auditory systems: A summary of audiometric and anatomical data and its implications for underwater acoustic impacts. NOAA Technical Memorandum NMFS-SWFSC-256:1-74.

Johnson, M.P. and P.L. Tyack. 2003. A digital acoustic recording tag for measuring the response of wild marine mammals to

sound. IEEE Journal of Oceanic Engineering 28(1):3-12.

MMS (Minerals Management Service). 2006. Gulf of Mexico OCS oil and gas lease sales: 2007-2012. Western Planning Area sales 204, 207, 210, 215, and 218; Central Planning Area sales 205, 206, 208, 213, 216, and 222. Draft Environmental Impact Statement. Volume I: Chapters 1-8 and appendices. OCS EIS/EA MMS 2006-062. Technical report, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA, USA.

Mobley, Jr., J., S.S. Spitz, K.A. Forney, R. Grotefendt, and P.H. Forestell. 2000. Distribution and abundance of odontocete species in Hawaiian waters: Preliminary results of 1993-98 aerial surveys. NMFS Administrative Report LJ-00-14C. National Marine Fisheries Service, La Jolla, CA, USA.

Mullin, K., W. Hoggard, C. Roden, R. Lohoefener, C. Rogers, and B. Taggart. 1991. Cetaceans on the upper continental slope in the north-central Gulf of Mexico. OCS Study MMS 91-0027. Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA, USA.

Nishiwaki, M. 1962. Aerial photographs show sperm whales' interesting habits. Norsk Havalangst-Tidende 51:395-398.

NMFS (National Marine Fisheries Service). 2006. Draft recovery plan for the sperm whale (*Physeter macrocephalus*). National Marine Fisheries Service, Silver Spring, Maryland, USA.

Palacios, D.M. and B.R. Mate. 1996. Attack by false killer whales (*Pseudorca crassidens*) on sperm whales (*Physeter macrocephalus*) in the Galapagos Islands. Marine Mammal Science 12:582-587.

Patenaude, N.J., W.J. Richardson, M.A. Smultea, W.R. Koski, G.W. Miller, B. Würsig, and C.R. Greene, Jr. 2002. Aircraft sound and disturbance to bowhead and beluga whales during spring migration in the Alaskan Beaufort Sea. Marine Mammal Science 18:309-335.

Pitman, R.L., L.T. Balance, S.I. Mesnick, and S.J. Chivers. 2001. Killer whale predation on sperm whales: Observations and implications. Marine Mammal Science 17:494-507.

Rice, D.W. 1989. Sperm whale *Physeter macrocephalus* (Linnaeus, 1758). In: S. H. Ridgway and R. Harrison, eds. Handbook of Marine Mammals. Volume 4: River dolphins and the larger toothed whales. Academic Press, San Diego, CA, USA, p. 177-234.

Richardson, W.J. and B. Würsig. 1997. Influences of man-made noise and other human actions on cetacean behaviour. Marine and Freshwater Behaviour and Physiology 29:183-209.

Richardson, W.J., C.R. Greene, Jr., C.I. Malme, and D.H. Thomson. 1995. Marine Mammals and Noise. Academic Press, San Diego, CA, USA. 576p.

Richter, C.F., S.M. Dawson, and E. Slooten. 2003. Sperm whale watching off Kaikoura, New Zealand: Effects of current activities on surfacing and vocalisation patterns. Science for Conservation Report No. 219. Department of Conservation, Wellington, New Zealand.

Richter, C., S. Dawson, and E. Slooten. 2006. Impacts of commercial whale watching on male sperm whales at Kaikoura, New Zealand. Marine Mammal Science 22:46-63.

Smultea, M.A., T.R. Kieckhefer, and A.E. Bowles. 1995. Response of humpback whales to an observation aircraft as observed from shore near Kauai, Hawaii, 1994. Final Report for the 1994 Marine Mammal Research Program of the Acoustic Thermometry of Ocean Climate (ATOC) study. Prepared by the Bioacoustics Research Program of the Cornell Laboratory of Ornithology, Cornell University, Ithaca, NY, USA. 46 p.

Urick, R.J. 1972. Noise signature of an aircraft in level flight over a hydrophone in the sea. Journal of the Acoustical Society of America 52:993-999.

Weller, D.W., B. Würsig, H. Whitehead, J.C. Norris, S.K. Lynn, R.W. Davis, N. Clauss, and P. Brown. 1996. Observations of an interaction between sperm whales and short-finned pilot whales in the Gulf of Mexico. Marine Mammal Science 12:588-594.

Whitehead, H. 2003. Sperm Whales: Social Evolution in the Ocean. University of Chicago Press, Chicago, IL, USA, 464 p.

Würsig, B., S.K. Lynn, T.A. Jefferson, and K.D. Mullin. 1998. Behaviour of cetaceans in the northern Gulf of Mexico relative to survey ships and aircraft. Aquatic Mammals 24:41-50.

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SHORT COMMUNICATION

OCCURRENCE OF LARVAL AND JUVENILE FISH IN MANGROVE HABITATS IN THE SIAN KA'AN BIOSPHERE RESERVE, QUINTANA ROO, MEXICO

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Introduction

Mangrove forests are ubiquitous in low lying coastal areas of tropical and subtropical zones of the world, including the lagoons of the Sian Ka'an Biosphere Reserve, Quintana Roo, Mexico. Mangroves are habitat for juvenile fish of both oceanic and estuarine origin (Vásquez-Yoemans 1992, Vásquez-Yoemans et al. 1992, Laegdsgaard and Johnson 1995). Development of the Caribbean coast of Mexico north and south of the Sian Ka'an Reserve is in large part focused on tourism-related endeavors such as destination resorts, scuba diving and fishing. While some of the development is innocuous, land acquisition for development of resorts has fragmented mangrove habitats in the region and likely altered their function. It has been shown in other mangrove estuaries that habitat fragmentation negatively impacts fish assemblages (Layman et al. 2004). Because of the importance of mangrove estuaries as juvenile fish habitat, loss of mangrove habitat may result in noticeable effects on adult recruitment to fisheries in tropical regions. Very little is known about the composition of larval and juvenile fish communities within the reserve.

Ichthyofaunal surveys of mangrove-lined estuaries worldwide have shown broadly similar taxonomic composition, including Eleotridae (sleepers), Gerreidae (mojarras), Mugilidae (mullets), Poeciliidae (livebearers), Gobiidae (gobies), Clupeidae (herrings) and Belonidae (needlefish) (Austin 1971, Blaber et al. 1989, Wright 1986, Thayer et al. 1987, Yáñez-Arancibia et al. 1988, Chong et al. 1990, Vásquez-Yoemans 1992; Vásquez-Yoemans and González 1992). In this research, we describe the juvenile fish community of two connected mangrove lagoons within the Sian Ka'an Biosphere Reserve at the end of the dry season (May).

Study Area

The Sian Ka'an Biosphere Reserve includes two bays, Bahía de la Ascensión and Bahía Espiritu Santo, and two shallow lagoons, Laguna Campechén, and Laguna Boca Paila (Figure 1). All sampling in this study took place in the shallow lagoon system. The two shallow water lagoons are created by a long narrow sand bar, are separated from the bay systems, and are connected to the Caribbean Sea through Boca Paila inlet. The lagoon system is about 1 m deep with deeper (2 – 3 m) channels, with fringing red mangrove (Rhizophora mangle), algal flats and seagrass beds. Shoal grass (Halodule sp.) was dense in the ocean pass (Boca Paila) and became sparse past the bridge and as the lagoon system extended inland.

Materials and Methods

Light trap sampling was conducted nightly from 7-20 May 1999, excluding 16 May 1999. Neuston net sampling was conducted during daylight hours on 9, 16 and 21 May 1999. Sampling included parts of two lunar cycles but not an entire cycle due to time constraints.

Three light trap sampling (LTS) stations were selected at increasing distances from Boca Paila inlet (Figure 1), but could not be spread too far due to the difficulties of navigating the lagoons at night. Microhabitats were red mangrove adjacent to seagrass beds (LTS 1), red mangrove adjacent to sandy bottom (LTS 2), and fringing red mangrove in a secondary channel (LTS 3). Light trap design followed that of Mueller et al. (1993). Electric diving lights (similar to cyalume sticks) were used as a light source. Two light traps were set at each site at sunset and retrieved about 1 h later. One trap was set within the red mangrove prop root complex and the second was located in the channel, about 1 m from the interior trap to ensure no overlap of illuminated areas. Upon retrieval, LTS samples were washed into 0.5 mm mesh biobags and fixed in 10% formalin overnight .

Four neuston net sampling sites were also selected at increasing distances from Boca Paila inlet and could be spread out further into the system since sampling occurred during the day (Figure 1). Microhabitats sampled were a secondary channel with sandy bottom fringed by red mangrove (NNS 1) rocky/sandy bottom adjacent to red mangrove, near a cenote (NNS 2), sandy bottom adjacent to red mangrove (NNS 3), and a seagrass bed adjacent to red mangrove (NNS 4). These sites were located in sufficient water depths (at least 1 m) to prevent the net from dragging the bottom. Neuston sampling was conducted with a 3:1 aspect ratio 60 cm net of 0.33

mm mesh. On each sampling day three net tows were conducted at each site and in haphazard directions for varying lengths of time. A high speed flowmeter was secured inside the neuston ring to calculate linear distance from the flowmeter coefficient. After each sampling effort, samples were washed down the net into the cod end (0.33 mm mesh) with buckets of seawater, and fixed in 10% formalin overnight.

Formalin-preserved samples from both gear types were transferred to 45% isopropyl alcohol for transportation and storage. Fish were identified to the lowest possible taxon and counted using keys in Ditty et al. (1994), Farooqi et al. (1995), Fritzsche (1978), Johnson (1978), Jones et al. (1978), Martin and Drewry (1978), Moser et al. (1984), Powles (1977), and Richards et al. (1994).

Raw data was standardized for both light trap and neuston net data. Mean set time was calculated for LTS samples, and total numbers of fish were then standardized to this unit of time for each sample (total fish/total minutes x mean minutes = standardized fish). NNS sampling was standardized to 100 m^3 (total fish/total volume x 100 m^3). Volume was calculated using the area of the neuston net ring multiplied by the distance sampled (volume = area x linear distance).

Results and Discussion

A total of 2,457 individuals representing 26 families and 50 species were collected during the sampling period. Light trap sampling captured 1,977 individuals from 42 species and 23 families (Table 1). The most commonly captured species in light traps were *Eucinostomus* spp., *Harengula jaguana*, *Cyprinodon variegatus* and an unidentified Gobiid. Neuston net sampling collected 480 individuals from 20 species and 13 families (Table 2), with the most commonly captured species including 2 unidentified goby species and an unidentified atherinid. Family dominance was similar between gears with Gobiidae, Atherinidae (silversides), and Gerreidae representing nearly 70% of the total catch in both. Other important families included Clupeidae and Cyprinodontidae (killifish) in light trap collections and Belonidae and Syngnathidae (pipefish) in the neuston net collections. Families exclusive to light trap collections were Balistidae (triggerfish), Lambrisomidae (lambrisomid blennies), Clupeidae, Eleotridae, Elopidae (ladyfish), Megalopidae (tarpon), Mugilidae, Ophichthidae (snake eels), Opistognathidae (jawfish), Pomacentridae (damselfish), Scaridae (parrotfish), Scorpaenidae (scorpionfish), Sparidae

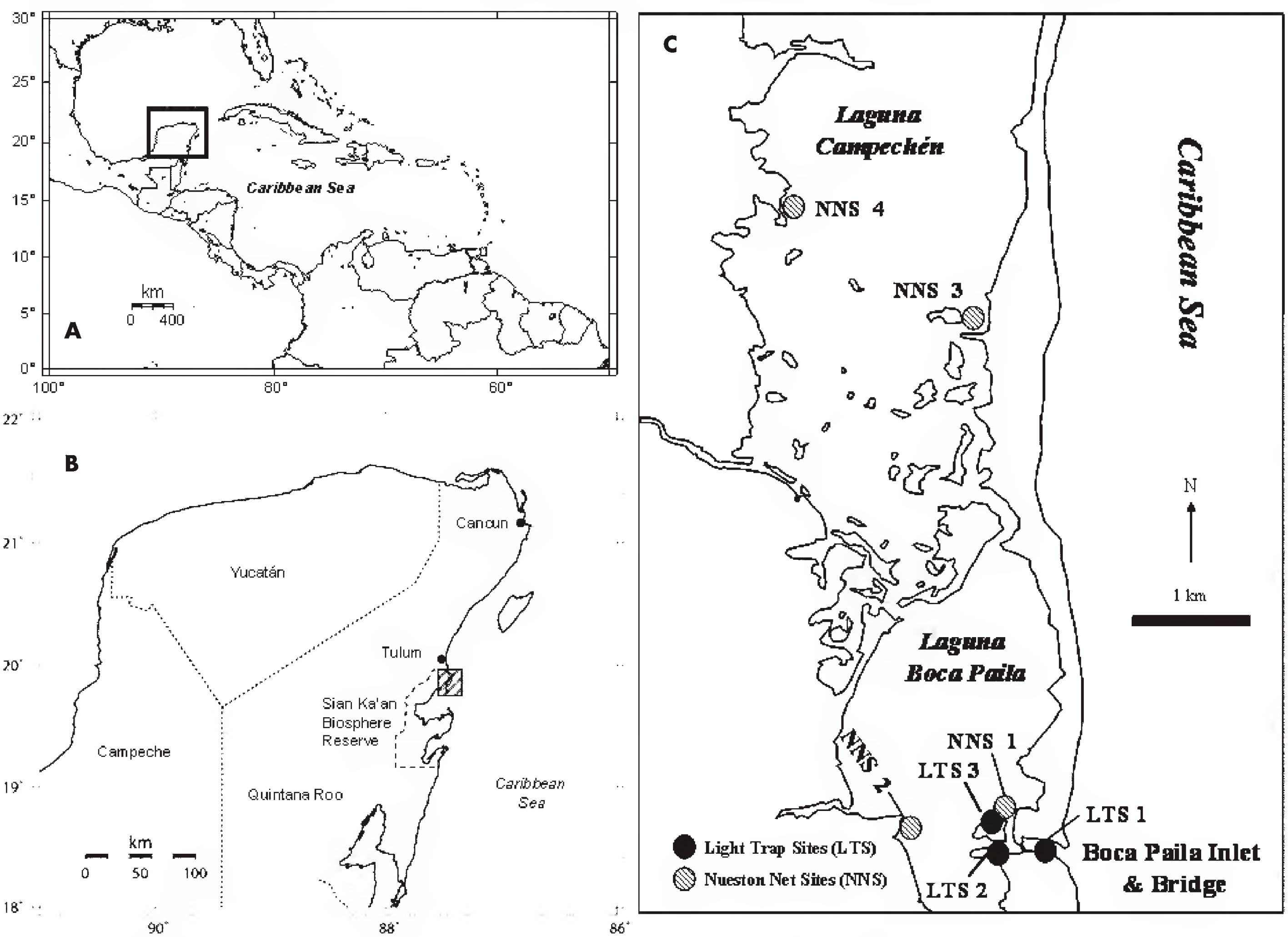

Figure 1. *Maps showing the location of the Sian Ka'an Biosphere Reserve on the Yucatán Peninsula, in Quintana Roo, Mexico (A), with detail of the study area (denoted by the hatched square; B) and location of sampling sites (C).*

TABLE 1. *Mean catch per unit effort (CPUE; fish/minute) of species in light trap stations, with standard error in parenthesis. Phylogenetic order, family and species names follow McEachran and Flechhelm (1998) and McEachran and Flechhelm (2005).*

	LTS 1	**LTS 2**	**LTS 3**		**LTS 1**	**LTS 2**	**LTS 3**
Elopidae				Carangidae			
Elops saurus	0.04 (0.06)	0.00	0.06 (0.06)	*Trachinotus falcatus*	0.04 (0.06)	0.00	0.00
Megalopidae				Gerreidae			
Megalops atlanticus	0.04 (0.06)	0.00	0.00	*Eucinostomus* spp.	17.21 (6.41)	10.12 (3.91)	9.64 (4.06)
Ophichthidiae				Sparidae			
Myrophis punctatus	0.04 (0.06)	0.03 (0.05)	0.00	Sparidae species A	0.00	0.2 (0.2)	0.00
Clupeidae				Pomacentridae			
Harengula jaguana	0.04 (0.06)	10.87 (10.22)	2.57 (1.98)	*Stegastes* spp.	0.00	0.03 (0.05)	0.00
Jenkinsia lamprotaenia	0.00	0.59 (0.49)	0.16 (0.22)	Scaridae			
Clupeidae species A	0.00	0.1 (0.1)	0.00	*Scarus vetula*	0.00	0.03 (0.05)	0.00
Engraulidae				Lambrisomidae			
Anchoa lamprotaenia	0.00	0.1 (0.11)	0.00	*Starksia lepicoelia*	0.18 (0.25)	0.00	0.00
Belonidae				Gobiesocideae			
Strongylura notata	0.44 (0.26)	0.14 (0.11)	0.06 (0.06)	*Gobiesox* spp.	0.49 (0.58)	0.00	0.00
Strongylura spp.	0.27 (0.16)	0.15 (0.1)	0.03 (0.04)	Gobiesocidae species A	0.09 (0.13)	0.03 (0.05)	0.00
Belonidae species A	0.09 (0.09)	0.00	0.00	Eleotridae			
Hemiramphidae				*Dormitator maculatus*	0.87(0.50)	0.13 (0.15)	0.06 (0.06)
Hyporhamphus unifasciatus	0.49 (0.35)	0.00	0.00	Gobiidae			
Mugilidae				Species A	5.85 (2.23	1.02 (0.78)	0.56 (0.67)
Mugil curema	0.00	0.00	0.03 (0.04)	Species B	0.9 (0.54)	0.14 (0.15)	0.03 (0.04)
Atherinidae				Species C	0.05 (0.07)	0.00	0.00
Atherinimorus sp.	0.18 (0.25)	0.11 (0.15)	0.00	Species D	1.42 (1.18)	0.2 (0.16)	0.00
Atherinomorus stipes	0.00	0.33 (0.18)	0.23 (0.23)	Species E	0.08 (0.08)	0.07 (0.07)	0.00
Atherinidae species A	1.29 (0.97)	1.4 (0.74)	0.21 (0.13)	Species F	0.36 (0.36)	0.00	0.00
Cyprinodontidae				Species G	0.04 (0.06)	0.00	0.00
Cyprinodon variegatus	3.84 (1.98)	0.32 (0.2)	0.04 (0.05)	Balistidae			
Syngnathidae				Balistidae species A	0.04 (0.06)	0.00	0.00
Syngnathus dunckeri	0.23 (0.13)	0.00	0.00	Monacathidae			
Syngnathus spp.	1.12 (1.20)	0.00	0.00	*Stephanolepis setifer*	0.04 (0.06)	0.00	0.00
Syngnathidae species A	0.00	0.23 (0.33)	0.00	Tetraodontidae			
Scorpaenidae				*Sphoeroides parvus*	0.09 (0.08)	0.00	0.00
Scorpaena plumieri	0.04 (0.06)	0.00	0.00	*Sphoeroides* spp.	0.04 (0.06)	0.00	0.00
Scorpaena spp.	0.09 (0.09)	0.00	0.00	Unidentified	0.00	0.03 (0.05)	0.00
Opistognathidae				**Overall**	**36.16 (7.9)**	**26.43 (12.95)**	**13.68 (5.0)**
Opistognathus spp.	0.00	0.05 (0.07)	0.00	**Species Richness**	**32**	**23**	**13**

(porgies) and Tetraodontidae (puffers). Families exclusive to neuston net collections were Blenniidae (combtooth blennies), Fistulariidae (cornetfish) and Haemulidae (grunts).

In light traps, both fish CPUE and species richness decreased as distance from the Boca Paila inlet increased (Table 1). Gerreids (*Eucinostomus* spp.) dominated the catch at all sites, and represented 48% of total catch. Gerreids are typically abundant along sandy shorelines, bays, and estuaries, a few species are found in freshwater, and they are important components of fish communities in many estuaries (Böhlke and Chaplin 1993). *Eucinostomus* spp. comprised the largest year-round population of juvenile fishes in saline mangroves bordered by coral reefs in Puerto Rico (Austin 1971). Gerreids were also the dominant species in mangrove-lined bays in the Ten Thousand Islands, Florida (Tabb and Manning 1961, Carter et al. 1973, Colby et al. 1985).

Clupeids represented 20% of the total light trap collection and were the second most abundant family. However, of the total clupeids collected at light trap sites, 77% were captured on 10 May 1999, and were either *Harengula jaguana*

TABLE 2. *Mean density (fish/100 m³) of species in neuston net stations, with standard error in parenthesis. Phylogenetic order, family and species names follow McEachran and Flechhelm (1998) and McEachran and Flechhelm (2005).*

	NNS 1	NNS 2	NNS 3	NNS 4
Engraulidae				
Engraulidae species A	0.00	0.23 (0.23)	0.00	0.75 (0.75)
Belonidae				
Strongylura spp.	0.29 (0.10)	0.00	0.00	0.15 (0.15)
Hemiramphidae				
Hyporhamphus unifasciatus	0.27 (0.09)	0.76 (0.40)	0.14 (0.14)	0.00
Fistulariidae				
Fistulariidae species A	0.00	0.00	0.00	0.10 (0.10)
Atherinidae				
Atherinomorus stipes	0.00	0.16 (0.16)	0.00	0.00
Atherinidae species A	1.76 (0.59)	7.71 (3.08)	1.06 (0.48)	1.95 (1.00)
Cyprinodontidae				
Cyprinodon variegatus	0.00	0.37 (0.37)	0.00	0.00
Syngnathidae				
Syngnathus spp.	0.21 (0.07)	0.25 (0.25)	0.80 (0.48)	1.59 (0.63)
Syngnathidae species A	0.00	0.00	0.00	1.58 (1.58)
Carangidae				
Oligoplites saurus	0.00	0.00	0.29 (0.19)	0.57 (0.45)
Gerreidae				
Eucinostomus spp.	0.21 (0.07)	0.16 (0.16)	0.00	1.8 (1.35)
Haemulidae				
Haemulon spp.	0.00	0.00	0.28 (0.28)	0.00
Blenniidae				
Blennius spp.	0.00	0.00	0.00	0.23 (0.23)
Blenniidae species A	0.00	0.28 (0.19)	0.37 (0.25)	2.56 (1.70)
Gobiesocidae				
Gobiesox spp.	0.27 (0.09)	0.12 (0.12)	0.00	0.00
Gobiesocidae species A	0.00	0.50 (0.50)	0.21 (0.21)	0.00
Gobiidae				
Gobiidae species D	0.00	0.20 (0.20)	0.92 (0.70)	0.26 (0.26)
Gobiidae species H	0.00	0.00	0.43 (0.31)	0.00
Gobiidae species I	2.49 (0.83)	4.53 (1.75)	24.46 (15.96)	4.45 (2.54)
Gobiidae species J	0.29 (0.10)	1.10 (0.73)	4.06 (3.13)	3.13 (1.86)
Unidentified	2.84 (0.95)	1.35 (0.78)	4.45 (3.65)	2.49 (1.0)
Yolk sac larvae	0.00	0.00	0.00	0.12 (0.12)
Overall	**8.63 (2.09)**	**17.72 (4.74)**	**37.46 (19.6)**	**21.73 (6.88)**
Species Richness	**8**	**13**	**11**	**13**

or *Jenkinsia lamprotaenia*. *Jenkinsia* sp. and *Harengula* sp. also dominated mangrove estuaries in Puerto Rico (Rooker et al. 1996) and Bahía de la Ascension (Vásquez-Yoemans 1992).

Gobiids were dominant in this study and are generally important components of shallow water communities. In this study, Gobiidae represented one-half of the abundance in neuston net collections, with 73% of individuals collected on the same day at NNS 1 and NNS 2. Elsewhere, gobiids dominated catch from shallow coral reef habitat at One Tree Island, Great Barrier Reef (Kingsford and Finn 1997), and in a lagoon reef at Moorea Island, French Polynesia (Dufour and Galzin 1993). In Florida (USA) estuaries, gobiids are important components of fish communities (Odum and Heald 1972). Gobiids were abundant and widely distributed in Bahía de la Ascension (Vásquez-Yeomans 1992) and represented 8.5% of the Bahaman

shore-fish fauna. Worldwide, they form a significant element of the tropical fish faunas (Böhlke and Chaplin 1993).

Due to the spatial and temporal limitations of the study the role of microhabitats in the lagoons could not be easily examined. However, larval and juvenile fish diversity and abundance within the mangrove estuary may have been enhanced by habitat contiguity. For example, species richness and CPUE were greater at LTS 1, where mangroves grew adjacent to dense seagrass beds, than it was in other LTS sites where mangroves were adjacent to bare bottoms. In contrast, the greatest density of fishes at neuston net sites was collected where the bottom was sandy (NNS 3), but species richness was greatest where there was a sandy/ rocky bottom and a cenote (NNS 2) and seagrasses (NNS 4).

Mangrove estuaries are significant in supporting local and global species diversity, and tropical fisheries are highly dependent on their continued healthy functioning. Maintaining the productivity of the mangrove estuaries within the Sian Ka'an Biosphere Reserve should be considered a high priority. Further analysis of fish recruitment in Laguna Boca Paila and Laguna Campechén is recommended with amendments to the spatial and temporal components of experimental design. Future studies should correct sampling design shortfalls from this study: for example, sample all sites with both gears and on the same days. Effects of lunar periodicity, currents, tides, and on-shore wind on recruitment of larval fish also need to be investigated.

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Literature Cited

Austin, H.M. 1971. A survey of ichthyofauna of the mangroves of western Puerto Rico during December, 1967 – August, 1968. Caribbean Journal of Science 11:27-39.

Blaber, S.J.M., D.T. Brewer, and J.P. Salini. 1989. Species composition and biomasses of fish in different habitats of a tropical northern Australian estuary: their occurrence in the adjoining sea and estuarine dependence. Estuarine, Coastal and Shelf Science 29:509-531.

Böhlke, E.B. and C.C.G. Chaplin. 1993. Fishes of the Bahamas and adjacent tropical waters. University of Texas Press, Austin, TX, USA, 771 p.

Carter M.R., L.A. Burns, T.R. Cavinder, K.R. Duggen, P.C. Fore, D.B. Hicks, H.L. Revelles, and T.W. Schmidt. 1973. Ecosystem analysis of the Big Cypress swamp and estuaries. Ecological Report No. DI-SFEP-74-51. U.S. Environmental Protection Agency, Athens, GA, USA, 331 p.

Chong, V.C., A. Sasekumar, M.U.C. Leh, and R.D. Cruz. 1990. The fish and prawn communities of a Malaysian coastal mangrove system, with comparisons to adjacent mudflats and inshore waters. Estuarine, Coastal and Shelf Science 31:703-722.

Colby, D.R., G.W. Thayer, W.F. Hettler Jr., and D.S. Peters. 1985. A comparison of forage fish communities in relation to habitat parameters in Faka Union Bay, Florida and eight collateral bays during the wet season. NOAA Technical Report NMFS-SEFC–162, Southeast Fisheries Science Center, Miami, FL, USA, 87 p.

Ditty, J.G., E.D. Houde, and R.F. Shaw. 1994. Egg and larval development of Spanish sardine *Sardinella aurita* (Family: Clupeidae), with a synopsis of characters to identify clupeid larvae from the northern Gulf of Mexico. Bulletin of Marine Science 54:367-380.

Dufour, V. and R. Galzin. 1993. Colonization patterns of reef fish larvae to the lagoon at Moorea Island, French Polynesia. Marine Ecology Progress Series 102:143-152.

Farooqi, T., R.F. Shaw, and J.G. Ditty. 1995. Preliminary guide to the identification of the early life history stages of anchovies (Family: Engraulidae) of the western Atlantic. NOAA Technical Memorandum NMFS-SEFSC-358. Southeast Fisheries Science Center, Miami, FL, USA, 65 p.

Fritzsche, R.A. 1978. Development of fishes of the mid-Atlantic bight: an atlas of egg, larval, and juvenile stages, Chaetodontidae through Ophidiidae. Report No. FWS/OBS-78/12, Volume 5. U.S. Department of the Interior, Fish and Wildlife Service, Office of Biological Services Program, Ft. Collins, CO, USA, 340 p.

Johnson, G.D. 1978. Development of fishes of the mid-Atlantic bight: an atlas of egg, larval, and juvenile stages, Carangidae through Ephippidae. Report No. FWS/OBS-78/12, Volume 4. U.S. Department of the Interior, Fish and Wildlife Service, Office of Biological Services Program, Ft. Collins, CO, USA, 311 p.

Jones, P.W., F.D. Martin, and J.D. Hardy. 1978. Development of fishes of the mid-Atlantic bight: an atlas of egg, larvae, and juvenile stages, Acipenseridae through Ictaluridae. Report No. FWS/OBS-78/12, Volume 1. U.S. Department of the Interior, Fish and Wildlife Service, Office of Biological Services Program, Ft. Collins, CO, USA, 366 p.

Kingsford, M.J. and M. Finn. 1997. The influence of phase of the moon and physical processes on the input of presettlement fishes to coral reefs. Journal of Fish Biology 51:176-205.

Laegdsgaard, P. and C.R. Johnson. 1995. Mangrove habitats as nurseries: unique assemblages of juvenile fish in subtropical mangroves in eastern Australia. Marine Ecology Progress Series 126:67-81.

Layman, C.A., D.A. Arrington, R.B. Langerhans, and B.R. Silliman. 2004. Degree of fragmentation affects fish assemblage structure in Andros Island (Bahamas) estuaries. Caribbean Journal of Science 40:232-244.

Martin, F.D. and G.E. Drewry. 1978. Development of fishes of the mid-Atlantic bight: an atlas of egg, larval, and juvenile stages, Stromateidae through Ogcocephalidae. Report No. FWS/OBS-78/12, Volume 6. U.S. Department of the Interior, Fish and Wildlife Service, Office of Biological Services Program, Ft. Collins, CO, USA, 416 p.

McEachron, J. D. and J. D. Fechhelm. 1998. Fishes of the Gulf of Mexico, Volume I: Myxiniformes to Gasterosteiformes. University of Texas Press, Austin, TX, USA, 1112 p.

McEachron, J. D. and J. D. Fechhelm. 2005. Fishes of the Gulf of Mexico, Volume II: Scorpaeniformes to Tetraodontiformes. University of Texas Press, Austin, TX, USA, 1004 p.

Moser, H.G., W.J. Richards, D.M. Cohen, M.P. Fahay, A.W. Kendall, Jr., and S.L. Richardson (eds.). 1984. Ontogeny and Systematics of Fishes. American Society of Ichthyologists and Herpetologists Special Publication 1. Allen Press, Lawrence, KS, USA, 760 p.

Mueller, G., M. Horn, J. Kahl Jr., T. Burke, and P. Marsh. 1993. Use of larval light traps to capture razorback sucker (*Xyrauchen texanus*) in Lake Mohave, Arizona-Nevada. Southwestern Naturalist 38:399-402.

Odum, W.E. and E.J. Heald. 1972. Trophic analyses of an estuarine mangrove community. Bulletin of Marine Science 22:671-738.

Powles, H. 1977. Description of larval *Jenkinsia lamprotaenia* (Clupeidae, Dussumieriinae) and their distribution off Barbados, West Indies. Bulletin of Marine Science 27:788-801.

Richards, W.J., K.C. Lindeman, J.L. Shultz, J.M. Leis, A. Röpke, M.E. Clarke, and B.H. Comyns. 1994. Preliminary guide to the identification of the early life history stages of lutjanid fishes of the western central Atlantic. NOAA Technical Memorandum No. NMFS-SEFSC-345., Southeastern Fisheries Science Center, Miami, FL, USA, 49 p.

Rooker, J.R., G.D. Dennis, and D. Goulet. 1996. Sampling larval fishes with a nightlight lift-net in tropical inshore waters. Fisheries Research 26:1-15.

Tabb, D.C. and R.B. Manning. 1961. A checklist of the flora and fauna of northern Florida Bay and adjacent brackish waters of the Florida mainland collected during the period July, 1957 through September, 1960. Bulletin of Marine Science of the Gulf and Caribbean 11:552-649.

Thayer, G.W., D.R. Colby, and W.F. Hettler. 1987. Utilization of the red mangrove prop root habitat by fishes in south Florida. Marine Ecology Progress Series 35:25-38.

Vásquez-Yoemans, L. 1992. Larvas de peces de la Bahía de la Ascensión, Quintana Roo, México. In: D.L. Navarro and J.G. Robinson, eds. Diversidad Biológica de la Reserva de la Biosfera de Sian Ka'an, Quintana Roo, México. CIQRO, Chetumal, Quintana Roo, México, p. 321-330.

Vásquez-Yoemans, L. and M.A. González. 1992. Peces marinos de las costas de Quintana Roo, México. In: D.L. Navarro and J.G. Robinson, eds. Diversidad Biológica de la Reserva de la Biosfera de Sian Ka'an, Quintana Roo, México. CIQRO, Chetumal, Quintana Roo, México, p. 361- 373.

Vásquez-Yoemans, L., W.J. Richards, and M.A González. 1992. Fish larvae of Quintana Roo coastal and offshore waters. In: D.L. Navarro and J.G. Robinson, eds. Diversidad Biológica de la Reserva de la Biosfera de Sian Ka'an, Quintana Roo, México. CIQRO, Chetumal, Quintana Roo, México, p. 287-303.

Wright, J.M. 1986. The ecology of fish occurring in shallow water creeks of a Nigerian mangrove swamp. Journal of Fish Biology 29: 431-441.

Yáñez-Arancibia, A., A.L. Lara-Dominguez, J.L. Rojas-Galaviz, P. Sánchez-Gil, J.W. Day Jr., and C.J. Madden. 1988. Seasonal biomass and diversity of estuarine fishes coupled with tropical habitat heterogeneity (southern Gulf of Mexico). Journal of Fish Biology 33:191-200.

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SHORT COMMUNICATION

FIRST RECORD OF *CERATASPIS MONSTROSA*, A LARVAL OCEANIC PENAEOID CRUSTACEAN, FROM THE GULF OF MEXICO

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Introduction

The genus *Cerataspis* (Gray, 1838) (Crustacea, Decapoda), assigned to the Penaeoidea by Burkenroad (1936) and Heegaard (1966), has circumglobal distribution between 40°N and 40°S (Morgan et al. 1985) and is represented in the Atlantic Ocean by two rarely collected species, *C. monstrosa* and *C. petiti*. Descriptions of both species are based on the larval forms since their adult form remain undescribed (Morgan et al. 1985). Although the larval development of *Cerataspis* is described by five mysis stages (I-V) (Heegaard 1966), the large larva is very un-mysid like and appears more like that of a megalops with the abdomen bend slightly toward the thorax. Other early developmental stages and life history aspects of *Cerataspis* are unknown. The bulky shape of the spectacular carapace with its various tubercles, horns, spines and large oil droplets contained in four pair of dorsal carapace tubercles most likely provide buoyancy for this pelagic life stage (Heegaard 1966, Morgan et al. 1985). Although larval *Cerataspis*, particularly the last three mysis stages, are typically pelagic, Heegaard (1966) suggested the adult form might be a reptant penaeoid which lives in the abyssal zone.

Heegaard (1966) reported 41 specimens of *Cerataspis* (26 *C. monstrosa*; 15 *C. petiti*) by mysis larval stage. Nine of the *C. monstrosa* (six from plankton samples; three from stomach contents of dolphinfish (*Coryphaena* spp.) and 13 of the *C. petiti* (all from plankton collections) were reported from the Atlantic Ocean. Morgan et al. (1985) further provided analysis of an additional 240 specimens of *Cerataspis* collected during surveys conducted off the southeastern United States, including Batts (1972), Manooch et al. (1983), and Manooch and Mason (1984). One of those specimens was collected by plankton net, and all others were found in the stomach contents of either yellowfin tuna (*Thunnus albacares*), blackfin tuna (*Thunnus atlanticus*), skipjack tuna (*Katsuwonus pelamis*) or dolphinfish (*Coryphaena hippurus*). To date, assessment of plankton collections and food habits of pelagic fishes from the Atlantic Ocean, inclusive of the Gulf of Mexico (GOM) (Heegaard 1966, Morgan et al. 1985), has provided no records of *Cerataspis* from the GOM. We report the first record of *C. monstrosa* from the GOM, one collected by plankton net and two collected from the esophagus of a wahoo, *Acanthocybium solandri*.

Materials and Methods

A single specimen of *C. monstrosa* was collected from the GOM on 5 June 2004 at 1856 h by a Tucker trawl plankton net (1 m x 2 m, 0.333 mm mesh net) towed at 10m depth (Latitude 27° 18.7'N, Longitude 87° 28.5'W; GCRL Sta. 04011, Coll. 19-16). Water temperature and salinity at 10m were 27.8°C and 36.4 ppt., respectively.

Two specimens of *C. monstrosa* were removed from the esophagus of a wahoo at the Mississippi Gulf Coast Billfish Classic (Biloxi, Mississippi) fishing tournament on 5 June 1998. The 102 cm fork length and 13.4 kg total weight female wahoo was caught by surface trolled hook-and-line gear at 1530 h from the northern GOM (Latitude 28° 30' N, Longitude 86° 30' W) (Capt. Tripp Tolbert, pers. comm., Fort Walton Beach, Florida).

All specimens of *C. monstrosa* were photographed in fresh condition and fixed in 95% ethanol. Identification of *C. monstrosa* was based on morphometric characteristics of *Cerataspis* mysis larval stages (Heegaard 1966). The fixed specimens were measured (millimeters, mm) for carapace length (CL), carapace width (CW) and total length (TL) following Heegaard (1966) and Morgan et al. (1985).

Results and Discussion

The specimen of *C. monstrosa* collected by plankton net (Figure 1, GCRL Museum catalog number GCRL2512) measured 8.0 mm CL, 6.0 mm CW, and 15.0 mm TL, and was assigned mysis stage III. This specimen represents the second published record of the genus *Cerataspis* collected by plankton net from the western Central Atlantic Ocean. The first record (mysis stage I) was reported from a 1862 Danish expedition to the West Indies (Heegaard 1966). The specimen reported here was alive when removed from the net and quickly placed in a container with seawater for observation. The remarkable larva exhibited hues of pink, lavender and purple, coloration characteristic of many neustonic organisms (Morgan et al. 1985), and its movements

Figure 1. *Images of Cerataspis monstrosa (mysis stage III) collected in a plankton sample from the northern Gulf of Mexico, 5 June 2004.*
A. Ventro-lateral view; the bulbous, convoluted lateral surface partially obscures the four pair of dorsal carapace tubercles; scale bar = 5 mm. The specimen was straightened from its typical curved body shape to accommodate measurements.
B. Dorso-lateral view showing the dorsal carapace tubercles; scale bar = 3 mm.

(apparently the first recorded observations of live *Cerataspis*) can best be described as periodic, rapid flexing of the abdomen away from the thoracic region, followed by immediate resumption of the typical curved body shape. Movements were similar to the 'flipping' motion displayed by peneaid shrimp when removed from water. Unfortunately, it was not practical to rear the larva in the laboratory in an attempt to describe older developmental stages.

Mysis stage III *Cerataspis* most likely occur in surface waters (Morgan et al. 1985); however, the stage III specimen we reported here was collected at a depth of 10 m from the western boundary of the Loop Current. The Loop Current is characterized by convergences, upwellings and strong flow (current speed characteristically 50 cm/s) along its outer boundary where planktonic organisms, including larvae of Caribbean and southern GOM origins, can become entrained and transported into the northern GOM (Johnson et al. 1992, Gasca et al. 2001). The origin of the specimen collected from Loop Current waters, however, is unknown.

The two specimens of *C. monstrosa* collected from the esophagus of the wahoo (GCRL museum catalog number GCRL2513) measured 10.0 mm CL each, 6.0 and 7.0 mm CW, and 21.0 and 22.0 mm TL, respectively, and were assigned mysis stage IV. The specimens were collected ~6 h after capture of the wahoo, appeared in excellent condition, and displayed vivid coloration that suggested they were recent prey. Morgan et al. (1985) examined stomach contents records of 885 wahoo from the Atlantic Ocean and 1,315 from the Pacific Ocean and found no accounts of *Cerataspis*.

Magnuson and Heitz (1971) reported that volumes of pelagic crustaceans in the stomach contents of predatory fishes from the Pacific were inversely related to the mean gill raker gap, and that wahoo, which has no gill rakers, had no crustaceans among their stomach contents. Manooch and Hogarth (1983) suggested the absence of small items in the diet of wahoo was attributable to the lack of gill rakers. In contrast, Morgan et al. (1985) found stomach contents of pelagic fishes with small gill raker gaps (tunas) contained greater numbers of larval *Cerataspis* than stomach contents of pelagic fishes with large gill raker gaps (e.g., dolphin).

Morgan et al. (1985) reported that the pelagic macroalga *Sargassum* (Phaeophyceae) was found in the stomachs of yellowfin tuna which also contained *Cerataspis*, and further considered *Cerataspis* a member of the surface community associated with pelagic *Sargassum*. Pelagic *Sargassum*, recently designated as essential fish habitat (EFH) for wahoo (SAFMC 2003), accumulates in large mats and along oceanic

frontal boundaries in the northern GOM (Comyns et al. 2002). Unlike some pelagic fishes that primarily feed at or near the surface of the water, wahoo do not readily feed at the surface (Manooch and Hogarth 1983). Nevertheless, the wahoo with specimens of *C. monstrosa* reported here was caught on surface fishing gear towed adjacent to a *Sargassum* driftline (Capt. Tripp Tolbert, pers. comm., Fort Walton Beach, Florida) where the fish might have incidentally consumed the larvae.

Specimens of *C. monstrosa* reported here represent the first documented records of the genus *Cerataspis* from the GOM, as well as from wahoo. Furthermore, the specimen collected by plankton net represents the second published record of the genus *Cerataspis* collected by this method from the Western Central Atlantic Ocean. These collections expand the known range of *Cerataspis* to the GOM and contribute to the knowledge of penaeoid crustaceans in the wider Atlantic Ocean.

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Literature Cited

Batts, B.S. 1972. Food habits of the skipjack tuna, *Katsuwonus pelamis*, in North Carolina waters. Chesapeake Science 13:193-200.

Burkenroad. M.D. 1936. The Aristaeinae, Solenocerinae and pelagic Penaeinae of the Bingham Oceanographic Collection. Bulletin of the Bingham Oceanographic Collection, Yale University 5:1-151.

Comyns, B.H., N.M. Crochet, J.S. Franks, J.R. Hendon, and R.S. Waller. 2002. Preliminary assessment of the association of larval fishes with pelagic *Sargassum* habitat and convergence zones in the northcentral Gulf of Mexico. Proceedings of the Gulf and Caribbean Fisheries Institute 53:636-645.

Gasca, R., I. Castellanos, and D.C. Biggs. 2001. Euphausiids (Crustacea, Euphausiacea) and summer mesoscale features in the Gulf of Mexico. Bulletin of Marine Science 68:397-408.

Heegaard, P. 1966. Larvae of decapod Crustacea. The oceanic penaeids: *Solenocera –Cerataspis - Certaspides*. Carlsberg Foundation, Dana Report No. 67:1-147.

Johnson, D.R., J.D. Thompson, and J.D. Hawkins. 1992. Circulation in the Gulf of Mexico from Geosat altimetry during 1985-1986. Journal of Geophysical Research 97:2201-2214.

Magnuson, J.J. and J.G. Heitz. 1971. Gill raker apparatus and food selectivity among mackerels, tunas and dolphins. Fishery Bulletin 69:361-370.

Manooch, C.S., III and W.T. Hogarth. 1983. Stomach contents and giant trematodes from wahoo, *Acanthocybium solandri*, collected along the South Atlantic and Gulf coasts of the United States. Bulletin of Marine Science 33:227-238.

Manooch, C.S. and D.L. Mason. 1984. Comparative food studies of yellowfin tuna, *Thunnus albacares*, and blackfin tuna, *Thunnus atlanticus*, collected along southeastern and Gulf coasts of the United States. Brimleyana 11:33-52.

Manooch, C.S., D.L. Mason, and R.S. Nelson. 1983. Food and gastrointestinal parasites of dolphin *Coryphaena hippurus* collected along the southeastern and Gulf coasts of the United States. NOAA Technical Memorandum 124:36p.

Morgan, S.G., C.S. Manooch, III, D.L. Mason, and J.W. Goy. 1985. Pelagic fish predation on *Cerataspis*, a rare larval genus of oceanic penaeoids. Bulletin of Marine Science 36:249-259.

SAFMC. 2003. Fishery management plan for the dolphin and wahoo fishery of the Atlantic. South Atlantic Fishery Management Council in cooperation with the New England Fishery Management Council and Mid-Atlantic Fishery Management Council. South Atlantic Fishery Management Council, Charleston, SC, USA, 309 p.

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Documentation of a Gulf Sturgeon Spawning Site on the Yellow River, Alabama, USA

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SHORT COMMUNICATION

DOCUMENTATION OF A GULF STURGEON SPAWNING SITE ON THE YELLOW RIVER, ALABAMA, USA

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Introduction

Gulf sturgeon (*Acipenser oxyrinchus desotoi*) have experienced population declines over the past century, primarily due to the effects of impoundments (e.g., barriers and altered flow regimes), water quality degradation, and overfishing (USFWS, GSMFC and NMFS 1995). These declines prompted the listing of Gulf sturgeon as a threatened species under the Endangered Species Act (U.S. Fish and Wildlife Service 1991). Effective management of Gulf sturgeon requires knowledge of its life history. Considerable effort has been expended in documenting the timing of the freshwater spawning migration and the location of habitats critical for successful spawning. To date, spawning sites have been identified and confirmed by the collection of fertilized eggs in the Apalachicola (Wooley and Crateau 1982, Wooley et al. 1982), Choctawhatchee (Fox et al. 2000), Escambia (Craft et al. 2001), Pascagoula (Heise et al. 2004) and Suwannee rivers (Marchant and Shutters 1996, Sulak and Clugston 1998, 1999). However, the timing and location of Gulf sturgeon spawning in other river systems is still unknown.

The Yellow River originates in southeast Alabama and flows southwest across northwest Florida into Blackwater Bay, near Milton, Florida. Craft et al. (2001) documented Gulf sturgeon movement in the Yellow River, and Berg et al. (2007) was able to provide a population estimate in 2003 of 911 fish (95% confidence interval, 550-1550). The fact that adults move up into the Yellow River and that three young-of- the-year sturgeon have been collected there (Berg 2004) suggests that spawning takes place, although this had never been verified by the collection of eggs. To afford Gulf sturgeon in the system some protection, the Yellow River from Alabama State Highway 55 downstream to its discharge in Blackwater Bay has been designated as Gulf sturgeon critical habitat (USFWS and NOAA 2003).

The Gulf Sturgeon Recovery Plan (USFWS, GSMFC and NMFS 1995) stressed the need to provide maximum protection to Gulf sturgeon spawning habitat. The approach employed by various Gulf sturgeon researchers, including ourselves, to document spawning has been to identify potential spawning habitat on the basis of physical characteristics and/or tracking data, collect eggs, and then raise the eggs in the laboratory until the point where the larval fish can be identified (e.g., Marchant and Shutters 1996, Sulak and Clugston 1998, 1999). However, collecting eggs in any appreciable number is usually difficult, and these eggs may not always be viable upon return to the laboratory. Molecular methods provide an alternative means of identifying the species represented by an egg. Notable examples related to sturgeon conservation include cases where molecular markers were used to verify the sources of commercially available caviar (DeSalle and Birstein 1996, Birstein et al. 1999).

Parauka and Giorgianni (2002) reported that potential Gulf sturgeon spawning habitat is present in the Yellow River; however, efforts to document spawning by the collection of eggs or larvae have been unsuccessful in the past. Herein, we report on the first successful collection of eggs from a potential spawning site on the Yellow River and the verification of their identity as Gulf sturgeon by using molecular methods.

Materials and Methods

Field Methods

Egg samplers (Marchant and Shutters 1996, Sulak and Clugston 1998) consisted of a red circular floor-buffing pad (55.9 cm or 68.6 cm diameter) anchored to the bottom with a rebar grapnel hook. A 6.4 m length of braided polyrope (0.63 cm diameter) was attached to the pad, and a float was tied to the end of the line to mark the location of the pad and facilitate retrieval. Samplers were georeferenced at the time of deployment with a hand-held GPS. Sixty to seventy-seven pads were deployed from 11 April to 13 May 2005 at five potential Gulf sturgeon spawning sites in the Yellow River (60-65 pads at three sites between rkm 123-134; 10-12 pads at two sites between rkm 95-100, Figure 1). Parauka and Giorgianni (2002) identified these sites as having the limestone, cobble, and hard substrate composition characterized by other researchers as being associated with documented Gulf sturgeon spawning (Marchant and Shutters 1996, Sulak and Clugston 1998, 1999, Fox et al. 2000,

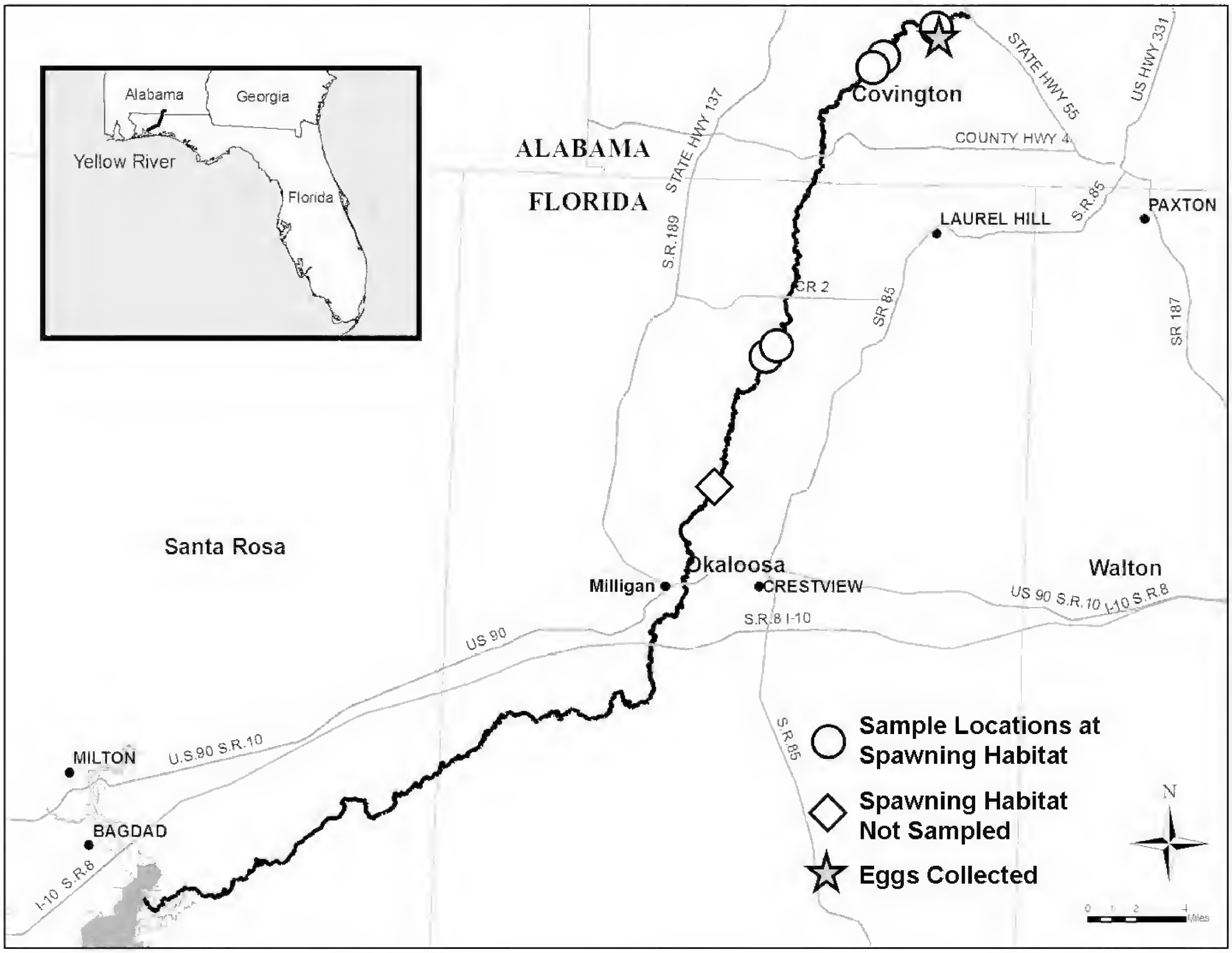

Figure 1.
Potential Gulf sturgeon spawning sites on the Yellow River, Alabama-Florida, USA, sampled during this study.

Craft et al. 2001). Samplers were retrieved and examined every 48-72 h for the presence of Gulf sturgeon eggs. Water temperature (°C), dissolved oxygen (mg/L) and conductivity (μS) measurements were taken 10 cm below the surface with a YSI-85 meter. River flow rates were obtained from the U.S. Geological Survey gauging station (02368000) located at rkm 64 on the Yellow River near Milligan, Florida. Any eggs collected from the pads were placed into water collected from the site and returned to the lab for rearing in covered, aerated holding containers. Eggs that failed to develop were placed into a tissue preservation buffer (Seutin et al. 1991).

The level of sampling effort and success in this study was compared to others by calculating catch per unit effort (CPUE). CPUE was quantified as the number of eggs collected divided by the number of sampling days (the number of sampling pads multiplied by the number of days the pads were deployed). Comparative data were gathered from the literature for the Pascagoula River (Heise et al. 2004) and from unpublished data for the Suwannee River (K. Sulak – United States Geological Survey). To account for the influence of prior knowledge of spawning locations, we calculated separate estimates of CPUE for the Pascagoula and Suwannee Rivers based on whether the sampling locations were known (targeted) or suspected (exploratory) spawning sites.

Molecular Methods

Each egg was rinsed with distilled water and blotted dry. Total genomic DNA was extracted from individual eggs with the DNeasy Tissue Kit, following the manufacturer's protocol (QIAGEN Inc., Valencia, CA). The quality of the extracted DNA was examined by gel electrophoresis on 1% agarose gels, stained with ethidium bromide (0.5 μg/ml) and then viewed under ultraviolet light. DNA concentrations were measured using a NanoDrop ND-100 spectrophotometer (NanoDrop Technologies Inc., Rockland, DE).

DNA from each egg was used in a polymerase chain reaction (PCR) to amplify a portion of the mitochondrial control region. We used the L15926 primer of Kocher et al. (1989) and another primer that we designed for Gulf sturgeon (GS-CRH1: 5'- GTGCCATTCACTGTTTGTCC). Amplifications were conducted in a total volume of 25 μl using 50 mM KCl, 10 mM Tris-HCl (pH 8.3), 0.01% gelatin, 2 mM $MgCl_2$, 200 μM dNTPs, 0.75 units *Taq* polymerase,

0.3 μM of each primer, 22.5-32 ng template DNA and water to the final volume. PCR cycling conditions consisted of an initial 1 min denaturing step at 95 °C followed by 30 cycles of 1 min at 95 °C, 1 min at 50 °C and 1 min at 72 °C. A final elongation step of 7 min at 72 °C completed the cycle. Several precautions were taken to avoid amplifying potential contaminants. We used fresh reagents, including buffers, $MgCl_2$, *Taq* polymerase and primers for the egg amplifications, and negative controls were employed for each set of reactions. Multiple independent reactions were also conducted on each of the egg samples.

Successful amplifications were prepared for sequencing using the ExoSAP-IT system (USB Co., Cleveland, OH). Both the forward and reverse strands of the control region fragment were cycle sequenced using a BigDye Terminator cycle sequencing kit v. 1.1 (Applied Biosystems, Foster City, CA). Sequencing reactions were performed with the GS-CRH1 primer and an internal primer reported by Ong et al. (1996) in lieu of the L15926 primer. Gel runs were performed at the Iowa State University DNA Sequencing and Synthesis Facility. Sequence data were aligned and edited with Sequencher 4.1 (Gene Codes Co., Ann Arbor, MI).

Two approaches were employed to identify the species represented by the eggs. First, we obtained comparative control region sequences from 14 adult Gulf sturgeon collected from rivers across the Gulf of Mexico. These individuals are part of the large and growing collection of tissue samples accumulated through the efforts of the many researchers involved with the Gulf sturgeon recovery effort. Samples were from the Apalachicola (n = 1), Choctawhatchee (n = 3), Yellow (n = 3), Escambia (n = 1), Pascagoula (n = 2) and Pearl rivers (n = 4). Second, we compared the egg sequences to all available sequences on GenBank (http://www.ncbi.nlm.nih.gov/) by nucleotide BLAST (Basic Local Alignment and Search Tool, Altschul et al. 1997, accessed June 6, 2005). The BLAST search returns a list of sequences matching the query sequence along with an *E*-value that measures the strength of the match. The *E*-value indicates how many other similar matches in the database would be expected by chance alone; thus the smaller the *E*-value the more likely the match between the query sequence represents real similarity.

DNA from each egg was also used to amplify the eight microsatellite loci used by Dugo et al. (2004) in their survey of population genetic structure in Gulf sturgeon. Precautions to avoid amplifying potential contaminants were made as previously described. All other methods and PCR reaction conditions followed Dugo et al. (2004) except for the use of 45-100 ng of DNA as template. Genotypes for each locus were compared to those obtained from adult Gulf sturgeon from across their range.

Results and Discussion

Spawning Site Features and Egg Collection

TABLE 1. *The numbers of each mitochondrial control region haplotype identified in adult Gulf sturgeon (labeled by the river of capture) and the one egg from the Yellow River that was successfully amplified and sequenced. Each haplotype is identified by an arbitrary letter code.*

	A	B	C	D	E
Apalachicola			1		
Choctawhatchee			2	1	
Yellow	1	2			
Escambia		1			
Pascagoula					2
Pearl		1	1		2
Egg	1				

Three eggs were collected at rkm 134 (known locally as "Dripping Rock") on 4 May 2005 (31°5'23.9"N, 86°27'31.9"W, Figure 1). The eggs were collected on two samplers about 100 m apart in water depths of 2.1 and 3.0 m, respectively, with an ambient water temperature of 18.1 °C, dissolved oxygen concentration of 7.06 mg/L, and conductivity of 40.2 μS. The river discharge was 105 m^3/s. The environmental parameters (especially temperature) and a rising hydrograph for the Yellow River during this period in 2005 matched the spawning conditions reported in other rivers (Chapman and Carr 1995, Sulak and Clugston 1998, Fox et al. 2000, Heise et al. 2004). Lunar cycles did not seem to influence spawning activity for Gulf sturgeon in the Yellow River, which was 5-6 d before the new moon and similar to what Fox et al. (2000) reported on the Choctawhatchee River.

The eggs were placed in water collected at the site and taken to the laboratory to allow development and hatch. Within 36 h, fungus appeared on the eggs, and they were removed from the receptacle and placed in the tissue preservation solution.

The CPUE for the five sites surveyed in this study averaged 0.001 eggs/sampling day. Exploratory sampling at four sites in the Pascagoula and three in the Suwannee produced an average CPUE of 0.003 and 0.011, respectively (Heise et al. 2004, K. Sulak – United States Geological Survey). Once spawning sites were confirmed, targeted egg sampling produced a higher CPUE for both the Pascagoula (0.119) and Suwannee (0.085).

Molecular Results

High molecular weight DNA was recovered from each of the three eggs with concentrations ranging from 15.0-21.4 ng/μL. Despite the presence of DNA in all three egg extractions, only one egg was successfully amplified in any of the PCR attempts. All adult samples were successfully amplified.

The amplified control region fragment was about 710 base pairs (bp) in length. Fourteen adults and two independent amplifications of the one egg were sequenced, producing 595 bp for each individual after editing. Five unique haplotypes

(A-E) were represented by the 14 adult Gulf sturgeon control region sequences (Table 1, GenBank accession numbers DQ088959-DQ088963). The number of nucleotide substitutions between haplotypes ranged from 1 to 12 and uncorrected p distances ranged from 0.0017 to 0.0202 (Table 2). The sequences from the egg were a perfect match with haplotype A, which was found in one adult Gulf sturgeon from the Yellow River. Haplotype B was only one base pair different and was found in the other two adults from the Yellow River, as well as one individual each from the Escambia and Pearl rivers.

Results of the BLAST query also supported the identification of the egg as belonging to Gulf sturgeon. Of the top 100 sequences producing significant alignments, the first 44 were from *A. oxyrinchus* (*A. o. oxyrinchus* and *A. o. desotoi*) with the remaining representing other *Acipenser* and *Huso* species. In all cases, the *E*-values were extremely small ($< 2.0 \times 10^{-64}$), signifying a strong match between the query and GenBank sequences.

Only four of eight microsatellite loci successfully amplified in the one egg sample. These loci included Aox-D32, Aox-D64, Aox-D242, and LS-68. The lack of a complete multilocus genotype for the egg precluded any formal analysis (e.g., assignment tests). However, the allele sizes for each locus successfully amplified matched those previously reported for Gulf sturgeon. Qualitative comparisons with the data of Dugo et al. (2004) revealed that the alleles present in the egg were also found in adult Gulf sturgeon collected from the Escambia and Yellow Rivers.

TABLE 2. *Number of pairwise base substitutions below the diagonal and uncorrected p distances above the diagonal between Gulf sturgeon mitochondrial control region haplotypes.*

	A	B	C	D	E
A	-	0.0017	0.0034	0.0034	0.0202
B	1	-	0.0017	0.0017	0.0185
C	2	1	-	0.0034	0.0168
D	2	1	2	-	0.0168
E	12	11	10	10	-

Implications

Our work verifies that Gulf sturgeon use the Yellow River for spawning and supports the importance of upper river hard bottom sites for successful reproduction (Sulak and Clugston 1998, 1999, Fox et al. 2000, Craft et al. 2001).

The documentation of spawning by Gulf sturgeon in the Yellow River highlights the need to protect the limited potential spawning habitat identified by Parauka and Giorgianni (2002). The eggs collected during this study, as well as three of the four young of the year Gulf sturgeon (Berg 2004, Ken Weathers, Alabama Department of Conservation and Natural Resources, pers comm. 2003) in previous studies, were from locations above a proposed dam site on the Yellow River. Additional studies are required to better characterize the habitats actually used by Gulf sturgeon in the Yellow River so that these river reaches can be better managed in accordance with the recovery actions described in the Gulf Sturgeon Recovery/Management Plan (USFWS, GSMFC and NMFS 1995).

Acknowledgments

The baseline collections and research for this project would not have been possible without the assistance and cooperation of the Gulf Sturgeon research community. In particular, the USGS provided the fundamental field logistics and baseline data that made this investigation possible. We extend our thanks to the field assistants associated with this work: M. Friedman (University of Florida), J. Russell (Southampton College and Long Island University), W. Harden (USGS), G. Yeargin (USGS) and A. Quaid (USGS). The fieldwork for this study was aided by volunteers M. Duncan of NOAA and B. Sawyer of Alabama Electric Cooperative. Personnel of the Conecuh National Forest assisted by providing campground space to be used as a base of operations during the project. K. Herrington, U.S. Fish and Wildlife Service, was responsible for putting together Figure 1. Helpful comments on the manuscript were provided by J. Powell and T. Slack, and invaluable editorial advice was provided by D. Kreiser.

Literature Cited

Altschul, S.F., T.L. Madden, A.A. Schäffer, J. Zhang, Z. Zhang, W. Miller, and D.J. Lipman. 1997. Gapped BLAST and PSI-BLAST: A new generation of protein database search programs. Nucleic Acids Research 25:3389-3402.

Berg, J.J. 2004. Population Assessment of the Gulf of Mexico Sturgeon in the Yellow River, Florida. M.S. Thesis. University of Florida, Gainesville, FL, USA, 82 p.

Berg, J.J., M.S. Allen, and K.J. Sulak. 2007. Population assessment of the Gulf of Mexico sturgeon in the Yellow River, Florida. American Fisheries Society Symposium 56:365-379.

Birstein, V.J., P. Doukakis, and R. DeSalle. 1999. Molecular phylogeny of Acipenserinae and black caviar species identification. Journal of Applied Ichthyology 15:12-16.

Carr, S.H., F. Tatman, and F.A. Chapman. 1996. Observations on the natural history of the Gulf of Mexico sturgeon (*Acipenser oxyrinchus desotoi* Vladykov 1955) in the Suwannee

River, southeastern United States. Ecology of Freshwater Fish 5:169-174.

Chapman, F.A. and S.H. Carr. 1995. Implications of early life stages in the natural history of the Gulf of Mexico sturgeon, *Acipenser oxyrinchus desotoi.* Environmental Biology of Fishes 43:407-413.

Craft, N.M., B. Russell, and S. Travis. 2001. Identification of Gulf sturgeon spawning habitats and migratory patterns in the Yellow and Escambia River systems. Final Report. Florida Department of Environmental Protection, Florida Marine Research Institute, and Fish and Wildlife Conservation Commission, St. Petersburg, FL, USA, 19 p.

DeSalle, R. and V.J. Birstein. 1996. PCR identification of black caviar. Nature 381:197-198.

Dugo, M.A., B.R. Kreiser, S.T. Ross, W.T. Slack, R.J. Heise, and B.R. Bowen. 2004. Conservation and management implications of fine-scale genetic structure of Gulf sturgeon in the Pascagoula River, Mississippi. Journal of Applied Ichthyology 20:243-251.

Fox, D.A., J.E. Hightower, and F.M. Parauka. 2000. Gulf sturgeon migration and habitat use in the Choctawhatchee River system, Alabama-Florida. Transactions of the American Fisheries Society 129:811-826.

Heise, R.J., W.T. Slack, S.T. Ross, and M.A. Dugo. 2004. Spawning and associated movement patterns of Gulf sturgeon in the Pascagoula River drainage, Mississippi. Transactions of the American Fisheries Society 133:221-230.

Kocher, T.D., W.K. Thomas, A. Meyer, S.V. Edwards, S. Pääbo, F.X. Villablanca, and A.C. Wilson. 1989. Dynamics of mitochondrial DNA evolution in animals: amplification and sequencing with conserved primers. Proceedings of the National Academy of Science, USA 86:6196-6200.

Marchant, S.R. and M.K. Shutters. 1996. Artificial substrates collect Gulf sturgeon eggs. North American Journal of Fisheries Management 16:445-457.

Ong, T.-L., S. Stabile, I. Wirgin, and J.R. Waldman. 1996. Genetic divergence between *Acipenser oxyrinchus oxyrinchus* and *A. o. desotoi* as assessed by mitochondrial DNA sequencing analysis. Copeia 1996:464-469.

Parauka, F.M. and M. Giorgianni. 2002. Availability of Gulf sturgeon spawning habitat in northwest Florida and southeast Alabama river systems. Technical Report. United States Fish and Wildlife Service, Panama City, FL, USA, 77 p.

Seutin, G., B.N. White, and P.T. Boag. 1991. Preservation of avian blood and tissue samples for DNA analyses. Canadian Journal of Zoology 69:82-90.

Sulak, K.J. and J.P. Clugston. 1998. Early life history stages of the Gulf sturgeon in the Suwannee River, Florida. Transactions of the American Fisheries Society 127:758-771.

Sulak, K.J and J.P. Clugston. 1999. Recent advances in life history of Gulf of Mexico sturgeon *Acipenser oxyrinchus desotoi* in the Suwannee River, Florida, USA: a synopsis. Journal of Applied Ichthyology 15:116-128.

U.S. Fish and Wildlife Service. 1991. Endangered and threatened wildlife and plants; determination of threatened status for the Gulf sturgeon. Federal Register 56:49653-49658.

USFWS, GSMFC and NMFS (U.S. Fish and Wildlife Service, Gulf States Marine Fisheries Commission and National Marine Fisheries Service). 1995. Gulf sturgeon (*Acipenser oxyrinchus desotoi*) recovery/management plan. Atlanta, GA, USA, 170 p.

USFWS and NOAA (U.S. Fish and Wildlife Service and National Oceanic and Atmospheric Administration). 2003. Endangered and threatened wildlife and plants; designation of critical habitat for the Gulf sturgeon; final rule. Federal Register 68:13369-13494.

Wooley, C.M. and E.J. Crateau. 1982. Observations of Gulf of Mexico sturgeon (*Acipenser oxyrinchus desotoi*) in the Apalachicola River, Florida. Florida Scientist 45:244-248.

Wooley, C.M., P.A. Moon, and E.J. Crateau. 1982. A larval Gulf of Mexico sturgeon (*Acipenser oxyrinchus desotoi*) from the Apalachicola River, Florida. Northeast Gulf Science 5:57-58.

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Behavior of an Escolar *Lepidocybium flavobrunneum* in the Windward Passage as Determined by Popup Satellite Archival Tagging

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SHORT COMMUNICATION

BEHAVIOR OF AN ESCOLAR *LEPIDOCYBIUM FLAVOBRUNNEUM* IN THE WINDWARD PASSAGE AS DETERMINED BY POPUP SATELLITE ARCHIVAL TAGGING

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Introduction

The escolar, *Lepidocybium flavobrunneum* (Smith), is a large gempylid fish found world-wide in tropical and temperate pelagic waters. The body is a uniform dark brown, with a sinuous lateral line, 4-6 dorsal and anal finlets, one main and two accessory keels on the caudal peduncle, and large greenish eyes (Smith 1997, Nakamura and Parin 2001). Little is known about the basic ecology of escolar but it is believed to feed primarily on squid, crustaceans, and a broad variety of fishes (Nakamura and Parin 1993). Very little is known of vertical or horizontal movement patterns, although commercial pelagic longline fishery catch records and scientific surveys suggest that this species inhabits mesopelagic waters between 200-885 m during the day and migrates vertically into epipelagic waters at night (Nakamura and Parin 1993). More recent work with pelagic longline gear has demonstrated that the species is occasionally caught at depths of < 50 m during overnight sets (Kerstetter and Graves 2006a).

The Windward Passage (Figure 1) is a deep-water strait in the northern Caribbean between the Republic of Cuba on the west and the island of Hispaniola (Republic of Haiti) on the southeast. It is about 80 km wide and is over 1700 m deep within the central channel. The Windward Passage is also characterized by seasonal variation in water flow and stratified temperature regimes at depth (Gunn and Watts 1982). This location was the site of a seasonal United States-based pelagic longline fishery targeting large swordfish (Family: Xiphiidae) that occurred primarily during the winter and spring months from December through March (NMFS 1999). However, due to international boundary issues, the U.S. fleet has been prohibited from fishing in the Windward Passage since 2004. Anecdotal reports suggest that escolar caught in this area by the pelagic longline fishery were larger than those caught elsewhere (Captain A. Mercier, F/V *Kristin Lee* and Captain G. O'Neill, F/V *Carol Ann*, pers. comm.). In addition to being one of the more common species in the historical Windward Passage fishery[1], escolar has become an increasingly valuable incidental retained catch in other areas for the fleet as a whole.

In June 2003, fisheries research was conducted in the Windward Passage using a chartered commercial pelagic longline vessel (Rice and Snodgrass 2003). This paper describes the habitat use by an escolar in this location tagged with a pop-up satellite archival tag (PSAT) that remained attached to the fish for 14 d. Data recovered from the PSAT were used to directly document diel vertical migration and ambient temperature range for the first time in a mesopelagic teleost.

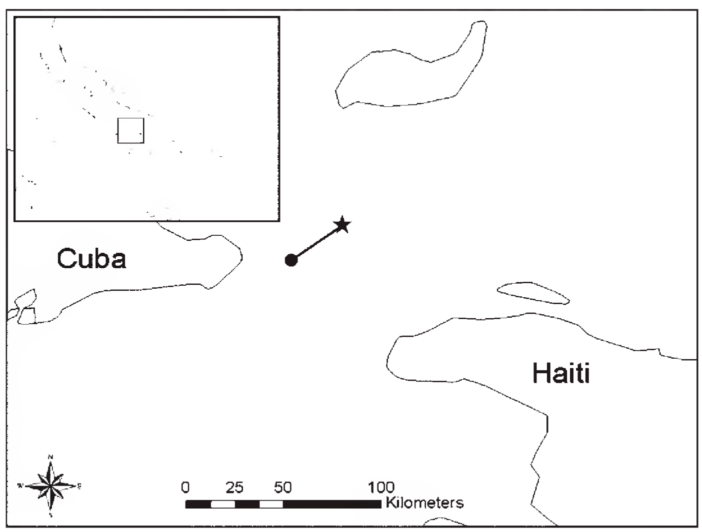

Figure 1.
Tagging (circle) and first transmission (star) locations for a PSAT tagged escolar in the Windward Passage, June 2003. The minimum straight-line distance between these two points was 44.4 km.

[1] NMFS Pelagic Observer Program data, 1992-2003. Available on-line at http://sefsc.noaa.gov/observerdata.jsp

Materials and Methods

Satellite tag and programming parameters

The PAT3 (Wildlife Computers, Redmond, WA) pop-up satellite archival tag was used for this deployment. The tag is cylindrical with a bulbous float on the non-attachment end and measures 38 cm by 4 cm (including antenna). This model records data on ambient pressure (depth, m), temperature (°C), and light level. Upon reaching its pre-programmed release date, the tag electronically detaches from the tether, floats to the surface, and transmits archived data summaries to the end user via the ARGOS satellite system. A small (about 1 cm length) cylindrical device called the "RD-1500" is also threaded onto the attachment tether between the tag and the animal. The device mechanically severs the tether if the tag reaches 1500 m, thereby preventing the tag from experiencing a fatal crush depth.

This tag model allowed the end user to pre-program a number of sampling intervals, as well as deployment duration. User-defined programming for this tag recorded ambient environmental data every 30 sec, which was then binned into each of the three daily histograms for transmission through the ARGOS satellite system. This version of PSAT segregated the sampled data into 12 intervals for both depth (0, 25, 50, 75, 100, 125, 150, 175, 200, 225, 250, and 1000 m) and temperature (12, 14, 16, 18, 20, 22, 24, 26, 28, 30, 32, and 60 °C). The programming then summarized these archived data in three discrete 8-h periods: 0100-0900, 0900-1700, and 1700-0100 GMT (2000-0400, 0400-1200, and 1200-2000 local time, respectively). In addition to these histograms, the tag also generated "pressure-depth-temperature" (PDT) profile data. This supplemental dataset provides additional summary data in which the depths and temperatures encountered by the animal are split into approximately equally-sized bins, as well as providing minimum and maximum depth and temperature for each 8-h binning period. The tag was programmed to detach from the fish on 12 August, a planned deployment duration of 59 d.

Tagging event and individual animal description

The fisheries research operations in the Windward Passage used monofilament pelagic longline gear ("Florida style" gear; see Berkeley et al. 1981). At about 0430 local time on 13 June 2003, a live escolar was observed during the haulback of the gear. It was hooked in the right side of its upper jaw with a size 18/0 circle hook baited with squid (*Illex* sp.). Data from an electronic hook time recorder (model HTR-1000; Lindgren-Pitman, Inc., Pompano Beach, FL, USA) indicated that the fish was hooked at 0159 local time, or about 2.5 h prior to observing the animal at boatside. Although not removed from the water for measurements, the captain and scientific crew both estimated the size of the escolar at about 122 cm and 27.2 kg. The fish was active, with intact musculature and eyes, and appeared pale brown in color.

The escolar was manually leadered to the gunwale of the vessel and retained in the water during the tagging procedure. The tag was rigged with a monofilament tether and nylon anchor using the basic attachment method described in Graves et al. (2002) and including a RD-1500 device. Using procedures similar to those described in Prince et al. (2002), the PSAT was inserted into the dorsal musculature, with the tether nylon anchor placed on the distal side of the dorsal pterygiophores. After the tags were attached, the hook was removed, and the fish was slowly resuscitated alongside the vessel for about five minutes prior to release. A GPS-based position of the vessel (20.19°N x 73.90°W), and sea surface temperature at the time of release (29.7°C) was recorded from the on-board vessel electronics.

Analyses

Because the tag was activated several hours prior to deployment, and the next period was incomplete, the first three 8-h periods were excluded from subsequent time-at-depth and time-at-temperature analyses. The last two 8-h periods were from the tag floating at or near the surface, and were similarly excluded. Moon phase and local times for sunrise, sunset, and nautical twilight were obtained from the U.S. Naval Observatory (http://aa.usno.navy.mil). A simple regression analysis was used to assess whether this animal ascended to shallower depths during nights of greater moonlight irradiance. All statistical tests were evaluated with an α-level of 0.05 using Statistical Analysis Software (SAS; v.9.0; SAS Institute, Cary, NC).

The ARGOS satellite system uses the Doppler shift in the tag transmissions to achieve estimates of varying accuracy on the geographic location of the tag. Horizontal displacement was calculated as the minimum straight-line distance between the location at which the animal was tagged and the first satellite transmission with an ARGOS location code corresponding to an accuracy of <1 km. This distance between start and end geographic position was calculated with the program INVERSE (NGS 2002, modified by M. Ortiz, NMFS SEFSC Miami Laboratory).

Results

The escolar PSAT prematurely released from the animal after 14 d at large. The first transmission to the ARGOS satellite system with the appropriate location code accuracy occurred on 26 June 2003 (20.36°N x 73.67°W; location accuracy 150-350 m). Using this position as the end point of the track, the animal moved a minimum straight-line distance during the deployment period of 44.4 km along a northeast heading (Figure 1). Over three days of multiple ARGOS satellite transmissions, 100% of the archived histograms and PDT profiles were recovered for analyses.

Immediately following tagging, the fish remained within the upper 25 m for more than 7.5 h of the first 8-h period before descending to depths > 250 m for the next 8-h period.

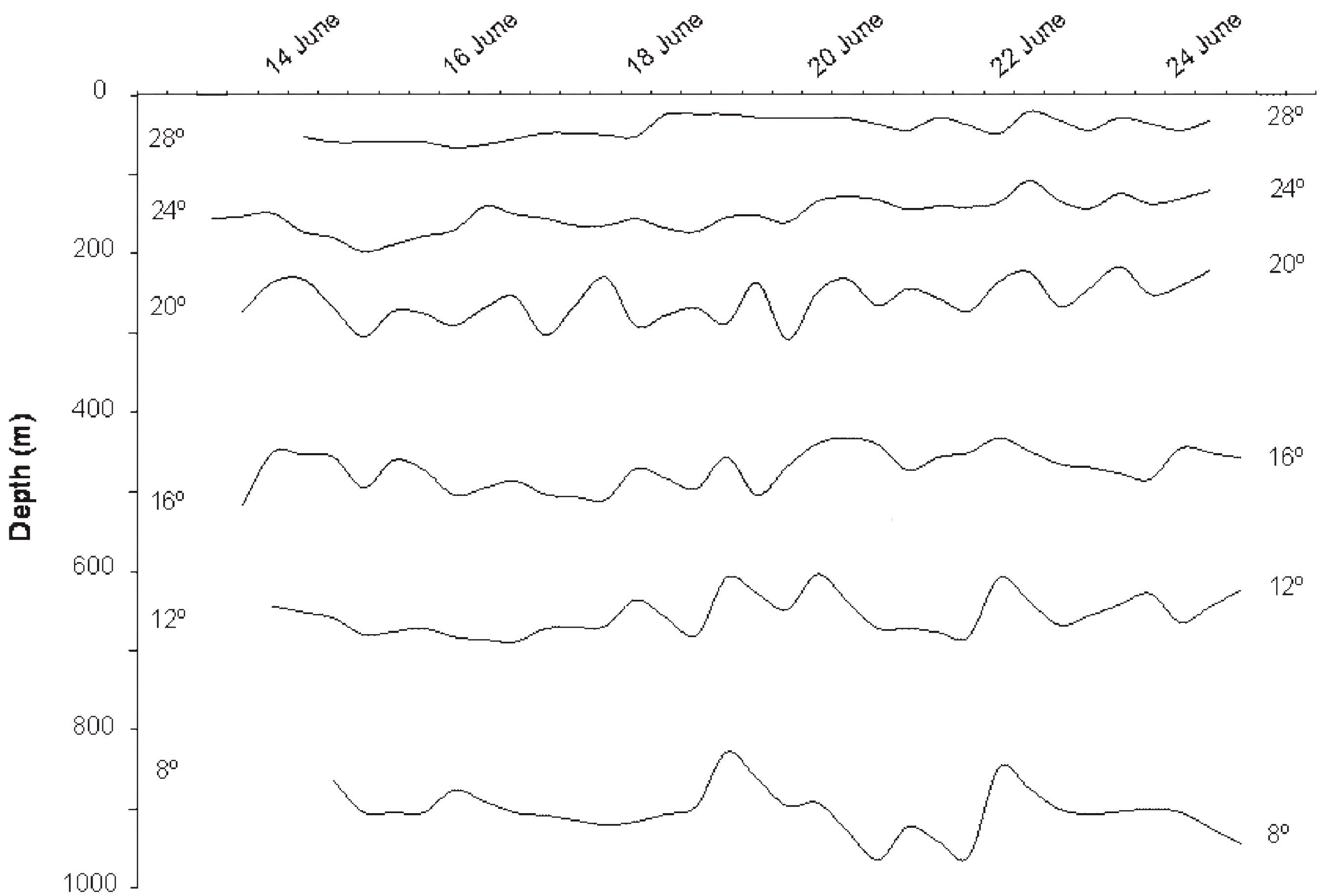

Figure 2.
Depth range of a PSAT tagged escolar in the Windward Passage by 8-h summary data bins. Dark grey bars indicate the "night" binning periods from the PSAT data, while light grey bars indicate the "day" and "crepuscular" periods (combined). Temperature stratifications at depth (thin lines) were generated from transmitted PDT data with a combination of recorded and interpolated temperature measurements.

Following this second period, the animal began a diel movement pattern for the remaining deployment length: during daylight hours the time-at-depth histograms indicate that the vast majority of time was spent at depths > 250 m, while the nighttime period was characterized by much shallower depth preferences (Figure 2). The 8-h crepuscular period included both light and dark photoperiods and had the broadest depth range, spanning from the shallow nighttime and deep daytime depth distributions (generally < 100 to > 800 m).

Although the unequal binning process in the tag programming precludes statistical comparisons between periods, the PDT profiles do allow for generalized reconstructions of the habitat preferences of the tagged animal. The PDT data provided depth maxima and minima for each 8-h binning period and allowed the reconstruction of both the inhabited water column and thermal layers using a combination of recorded and interpolated depth and temperature values (Figure 2). Overall depth and temperature use percentages were compared as day versus night periods and scaled to the proportion of total time over the course of the deployment period (Figure 3). The PDT data were also used to roughly reconstruct the vertical structure of the water column, which showed a relatively shallow mixed layer, followed by a weak thermocline that extended to about 250 m.

The U.S. Naval Observatory data indicate that sunrise in the Windward Passage during the tag deployment occurred at 0516 (± 2 min) and sunset at 1839 (± 2 min). The general agreement between programmed periods and local times of sunset and sunrise prompted an initial naming of period #1 as "night," period #2 as "crepuscular," and period #3 as "day." However, the similarity of the distributions of "crepuscular" and "day" periods prompted the collapse of both into one "day" period. Multiple comparisons of minimum temperatures (a proxy for maximum depth, which was not recorded) with the Tukey test for unequal sample sizes (Zar 1999) indicated significant differences between periods 1 and 2 and periods 1 and 3, but not periods 2 and 3. The three 8-h periods per day from the tag programming roughly corresponded to one night and two daylight photoperiods during the deployment, allowing for descriptive comparisons between the two periods.

The depth and temperature distributions from this

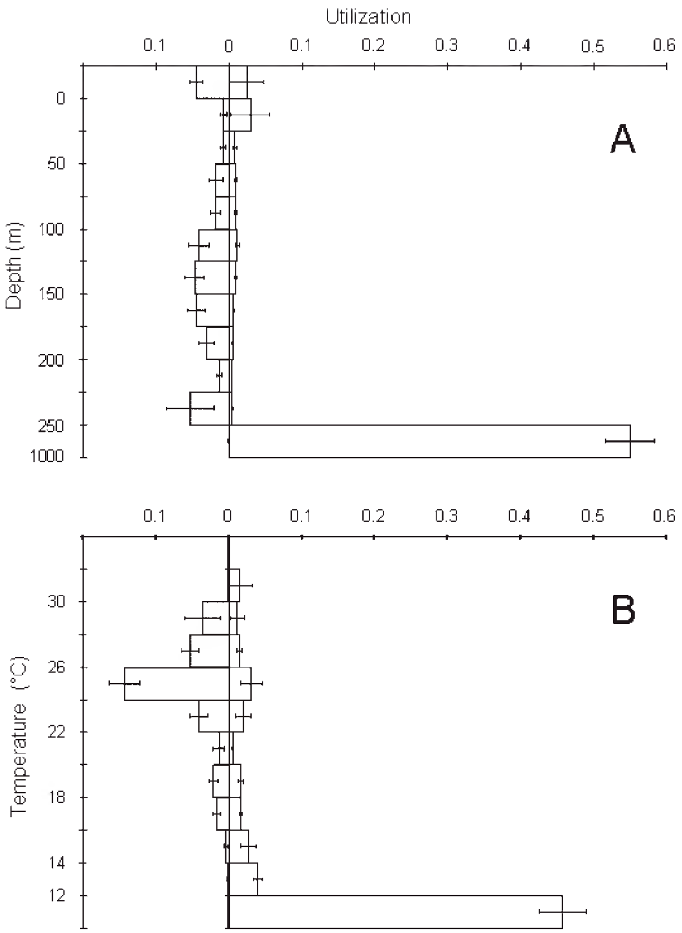

Figure 3.
Scaled habitat utilization of time of depth and temperature ($\bar{x}$ ± se) for a PSAT tagged escolar in the Windward Passage. Open bars are daylight (two 8-h summary periods per 24-h period) and dark bars are at night (one 8-h summary period per 24-h period). Due to the binning structure of the tag programming, depths between 250 and 1000 m were binned into the last depth category and temperatures < 12°C were similarly binned into the last temperature category.

fish show strong diel patterns in movement (Figure 3). The majority of the nighttime period (70%, ± 8.2% se) was spent above 150 m depth, while the majority of the daytime period (82.5%, ± 5.1% se) was spent below 250 m. Depth data from PDTs indicate that on at least one night, the escolar came within 5 m of the surface, although there was not a significant relationship between depth at night and available moonlight (r = -0.530, p = 0.093).

Temperature minima and maxima for each 8-h period were examined for temperature range and then grouped by 24-h period to obtain an estimate of total daily temperature range. The mean temperature range per 8-h period was 17.0°C (±0.73 se). Over the 11 deployment days with all three 8-h periods, the animal had a daily temperature range of 21.3°C (±0.38 se).

Discussion

Although PSAT technology has been successfully applied to other deep-diving animals (e.g., sperm whales, *Physeter macrocephalus*; Amano and Yoshioka 2003 and Humboldt or jumbo squid, *Dosidicus gigas*; Gilly et al. 2006), this tagging represents the first successful deployment of this technology for a mesopelagic teleost. The available data do not allow the determination of why the tag released prematurely, but additional research with tagging techniques on mesopelagic fishes may identify specific techniques that differ from those used with epipelagic animals, such as not including a RD-1500 device. As with most mesopelagic species, there is little available information on the behavior of escolar other than that based on incidental interactions with fisheries, such as the commercial pelagic longline fishery or the recreational swordfish fishery off the southeast coast of Florida. Through the use of this fisheries-independent PSAT technology, about 14 d of recorded behavior were obtained from an escolar for the first time.

This fish exhibited clear diel differences in depth preference. Unfortunately, the binning structure of our tag programming did not allow the reconstruction of short-duration movements within each pre-determined time period. For example, the reported time-at-depth histograms for this escolar could represent broad movements up to shallow depths, similar to the "U-shaped" depth record reported by Carey and Robison (1981) for tracked swordfish. Alternatively, the histograms could represent serial vertical movements, such as those seen in bigeye tuna *Thunnus obesus* (Musyl et al. 2003). Regardless of the vertical movement pattern these histograms actually represent, the data suggest that escolar do not remain within a single depth regime. Additional investigations may determine that a revision in the species depth distribution to "nictoepipelagic" is warranted.

The crepuscular period included the most time at depth of the three 8-h bins, rather than the daylight period, even though the sunrise and sunset times roughly corresponded with the bin start and end times. In Kerstetter and Graves (2006a), all of the escolar catches on hook-time recorders (HTRs) occurred at night or nautical twilight (dusk), suggesting that the species follows an isolume similar to other mesopelagic predators. HTRs during the present study similarly indicated that all escolar were captured between 2100 and 0500 local time (P.H. Rice, unpubl. data). Carey and Robison (1981) observed that swordfish frequently moved from depth to near-surface waters within a 1-h period.

It is likely that escolar follow the isolume, resulting in behavioral patterns similar to that demonstrated in swordfish. Data from the U.S. Naval Observatory indicate that nautical twilight for this location occurs about one hour prior to actual sunrise. The clear, oligotrophic waters of the northern Caribbean Sea and the absence of artificial light has been demonstrated to allow the transmission of

light to depth prior to sunrise (D.W. Kerstetter, unpubl. data), which may serve as a visual cue to an animal like the escolar that preferentially forages in low light-level conditions. The eyes of escolar are also large and very sensitive to light (E. Landgren, Lunds University, pers. comm.). Although the overlap in the programming of the tag allowed about 25% of the 8-h period to include nominal darkness, such short-duration use of shallow waters by a downward migrating fish following the isolume would presumably be masked by the larger percentage of time spent at depth.

The first data recording period of the track is shallower than subsequent identical time bins, and the next preceding period shows less depth variation than other periods during the same diel cycle. The disproportionately large size of the last bin in the tag programming might mask some of the intra-bin vertical movements, however, and PDT data indeed indicate a depth range during this second period between 272 and 784 m. It is unclear whether these first two time periods of the deployment were indicative of a so-called "recovery period" for the animal, similar to aberrant behaviors noted for other large pelagic fishes following tagging (e.g., Nelson 1990, Loefer et al. 2005, Kerstetter and Graves 2006b). Although briefly resuscitated prior to release, the fish was hooked for about 2.5 h, and little has been published on the time required for full physiological recovery for large pelagic fishes under various stress related conditions. This particular individual was also pale brown, rather than the common dark brown seen on other escolar at haulback, perhaps showing an additional sign of physiological stress.

The large water temperature extremes experienced by this animal on a daily basis are beyond those known for most fishes. The known exceptions all include species with some physiological mechanism to compensate for loss of body temperature at depth. Although Brill et al. (1999) and Brill and Lutcavage (2001) have suggested a maximum temperature range of about 8 °C for short-duration movements, the available summary data do not permit such an analysis for this individual escolar. Carey and Robison (1981) observed a temperature range of 19 °C in two hours for a tagged swordfish. Although similarly broad water-temperature ranges were observed for the tagged escolar, it is currently unknown what physiological mechanisms – if any – may be used by this species to allow for effective foraging at these temperature extremes.

As opposed to more epipelagic species such as tunas and istiophorid billfishes, very few mesopelagic fishes have been tagged with satellite tag technology. This paper presents new information on the movements and temperature preferences of an escolar based on one pop-up satellite archival tag deployment. The patterned movements to and from depth suggest that these behaviors were not abnormal, but instead likely represent regular, diel feeding migrations similar to those seen in swordfish. Further study of this species will improve our understanding of its biology and thermal adaptations.

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Literature Cited

Amano, M. and M. Yoshioka. 2003. Sperm whale diving behavior monitored using a suction-cup-attached TDR tag. Marine Ecology Progress Series 258:291-295.

Berkeley, S.A., E.W. Irby Jr., and J.W. Jolley Jr. 1981. Florida's commercial swordfish fishery: longline gear and methods. Florida Cooperative Extension Service, University of Miami; Florida Sea Grant Marine Advisory Bull. MAP-14. Miami, FL, USA, 23 p.

Brill, R.W., B.A. Block, C.H. Boggs, K.A. Bigelow, E.V. Freund, and D.J. Marcinek. 1999. Horizontal movements and depth distribution of large adult yellowfin tuna (*Thunnus albacares*) near the Hawaiian Islands, recorded using ultrasonic telemetry: implications for the physiological ecology of pelagic fishes. Marine Biology 133: 395-408.

Brill, R.W. and M.E. Lutcavage. 2001. Understanding environmental influences on movements and depth distributions of tunas and billfishes can significantly improve population assessments. American Fisheries Society Symposium 25:179-198.

Carey, F.G. and B.H. Robison. 1981. Daily patterns in the activities of swordfish, *Xiphias gladius*, observed by acoustic telemetry. Fishery Bulletin 79:277-291.

Gilly, W.F., U. Markaida, C.H. Baxler, B.A. Block, A. Boustany, L. Zeidberg, K. Reisenbichler, B. Robison, G. Bazzino, and C. Salinas. 2006. Vertical and horizontal migrations by jumbo squid, *Dosidicus gigas*, revealed by electronic tagging. Marine Ecology Progress Series 324:1-17.

Graves, J. E., B.E. Luckhurst, and E.D. Prince. 2002. An evaluation of popup satellite tags for estimating post release survival of blue marlin (*Makaira nigricans*) from a recreational fishery. Fishery Bulletin 100:134–142.

Gunn, J.T. and D.R. Watts. 1982. On the currents and water masses north of the Antilles/Bahamas arc. Journal of Marine Research 40:1-18.

Kerstetter, D.W. and J.E. Graves. 2006a. Effects of circle versus J-style hooks on target and non-target species in a pelagic longline fishery. Fisheries Research 80:239-250.

Kerstetter, D.W. and J.E. Graves. 2006b. Survival of white marlin (*Tetrapturus albidus*) released from commercial pelagic longline gear in the western North Atlantic. Fishery Bulletin104:434-444.

Loefer, J.K., G.R. Sedberry, and J.C. McGovern. 2005. Vertical movements of a shortfin mako in the western North Atlantic as determined by pop-up satellite tagging. Southeastern Naturalist 4:237-246.

Musyl, M.K., R.W. Brill, C.H. Boggs, D.S. Curran, T.K. Kazama, and M.P. Seki. 2003. Vertical movements of bigeye tuna (*Thunnus obesus*) associated with islands, buoys, and seamounts near the main Hawaiian Islands from archival tagging data. Fisheries Oceanography 12:152-169.

Nakamura, I. and N.V. Parin. 1993. FAO species catalogue. Vol. 15. Snake mackerels and cutlassfishes of the world (Families Gempylidae and Trichiuridae). FAO Fisheries Synopsis: Rome, Italy. 136 p.

NGS (National Geodetic Survey). 2002. Program INVERSE. Silver Spring, MD, USA,

National Marine Fisheries Service (NMFS). 1999. Final fishery management plan for Atlantic tunas, swordfish, and sharks. NOAA-NMFS-F/SF-Highly Migratory Species Management Division. Silver Spring, MD, USA, 1600 p.

Nelson, D.R. 1990. Telemetry studies of sharks: a review, with applications in resource management. In: H.L. Pratt, Jr., S.H. Gruber, and T. Taniuchi, eds. Elasmobranchs as living resources: advances in the biology, ecology, systematics, and the status of the fisheries. NOAA Technical Report, NOAA-NMFS-SWSC-90, La Jolla, CA, USA, p. 239-256.

Prince, E.D., M. Ortiz, A. Venizelos, and D.S. Rosenthal. 2002. In-water conventional tagging techniques developed by the cooperative tagging center for large, highly migratory species. American Fisheries Society Symposium 30:155–171.

Rice, P. H. and D. Snodgrass. 2003. Windward Passage scientific cruise report (2002-125): May 31–June 16, 2003. National Oceanic and Atmospheric Administration, Southeast Fisheries Science Center, Miami, FL, USA, 10 p.

Smith, C.L. 1997. National Audubon society field guide to tropical marine fishes of the Caribbean, the Gulf of Mexico, Florida, the Bahamas, and Bermuda. Alfred A. Knopf, Inc., New York, NY, USA, 720 p.

Zar, J.H. 1999. Biostatistical analysis, 4th edition. Prentice Hall, Upper Saddle River, NJ, USA, 663 p.

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Editorial

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EDITORIAL

As one famous Mississippi songwriter exclaims "Changes in latitudes, changes in attitudes, nothing appears quite the same" applies not only to life but to the *Gulf and Caribbean Research*. Since I became Editor-in-Chief starting with the 1997 issue, many changes and attitudes have changed regarding what the "Scope" of the journal should be (see *Gulf Research Reports,* volume 11 editorial), its scholastic and scientific value, and how it is managed and produced. We have come a long way from the first issue in 1961. Much of those changes were directly influenced by the expert guidance and energy of Sigrid "Dawne" Hard, who recently retired as Managing Editor of this journal. Dawne was integral to all aspects of this journal and brought it to a higher level of scholarship by her dedication to how the journal looked and her persistence on how it was edited. She could be a task-master but always with the goal of a better product. I miss her and her energy and talents and she made this journal better for all her dedicated work.

Luckily (for me) I was able to replace her with other talented and hard working professionals; Diana Reid (Graphic Designer), Angela R. Bone (Administrative Assistant) and Nancy J. Brown-Peterson (Assistant Editor) who share the duties of layout of manuscripts, tracking manuscripts, and editing for style and format, respectively. We are also lucky to continue to have Joyce M. Shaw (Librarian) to assist us with literature accuracy issues. We have made this transition with limited difficulties and have made some additional modifications in the journal. We have changed the format and style to make it more contemporary, have changed the cover to better reflect our "Scope," and now have additional early editing making the final product more consistent and accurate. All of this has allowed me to better focus on manuscript quality for the journal.

As we continue to evolve at *Gulf and Caribbean Research* our gratitude is extended to Dr. William Hawkins (GCRL Director) for his guidance and support of this 47 year old journal!

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